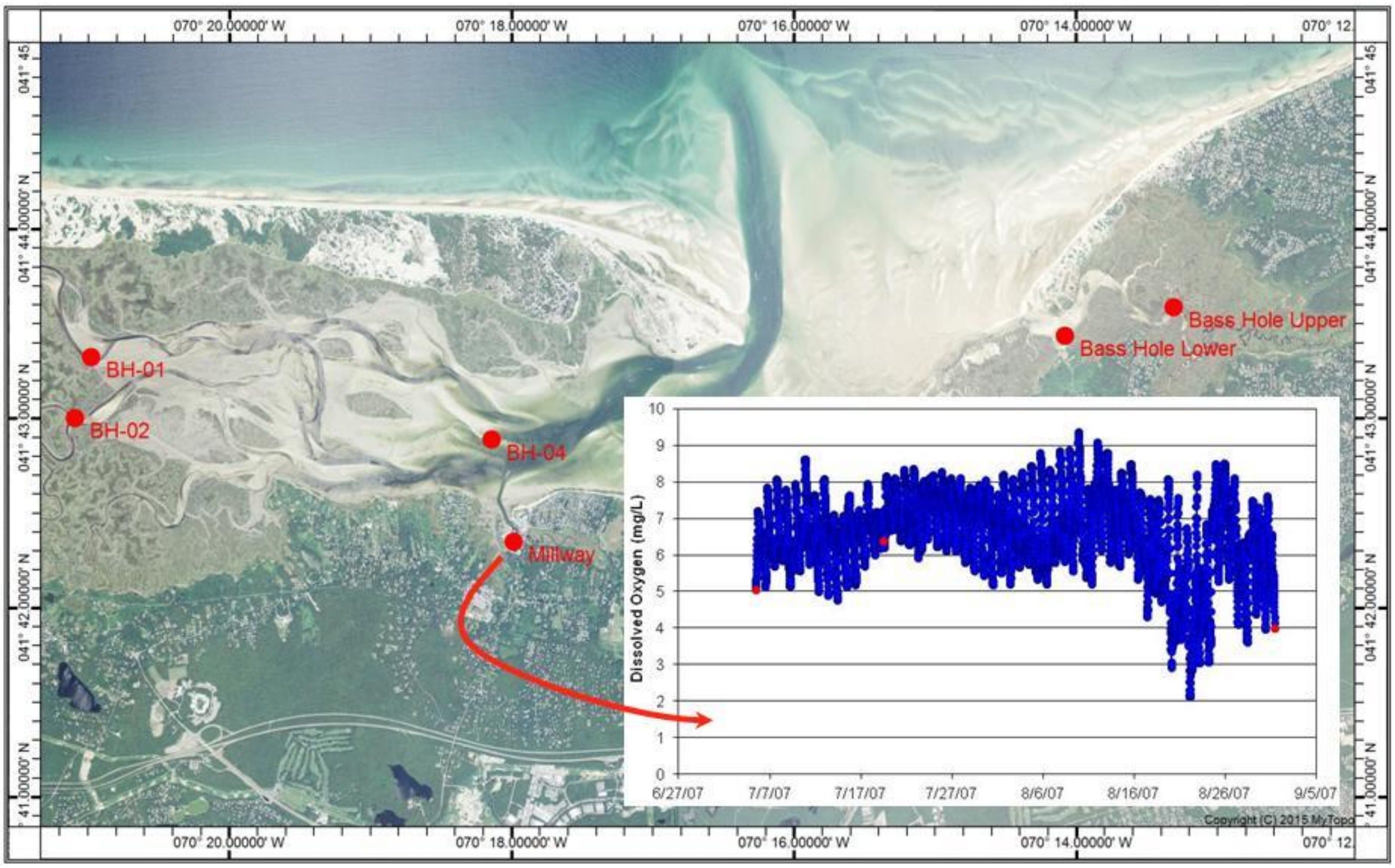
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Threshold for the Barnstable Great Marshes-Bass Hole Estuarine System Town of Barnstable & Dennis, Massachusetts



University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of
Environmental Protection

DRAFT REPORT – June 2017

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Threshold for the Barnstable Great Marshes -Bass Hole Estuarine System Town of Barnstable & Dennis, Massachusetts

DRAFT REPORT – June 2017

Brian Howes
Roland Samimy
Ed Eichner
David Schlezinger

Trey Ruthven
John Ramsey

Phil "Jay" Detjens

Contributors:

US Geological Survey
Don Walters and John Masterson

Applied Coastal Research and Engineering, Inc.
Elizabeth Hunt and Sean Kelley

Massachusetts Department of Environmental Protection
Charles Costello and Brian Dudley (DEP project manager)

SMAST Coastal Systems Program
Jennifer Benson, Michael Bartlett, Sara Sampieri

Cape Cod Commission
Tom Cambareri

ACKNOWLEDGMENTS

The Massachusetts Estuaries Project Technical Team would like to acknowledge the contributions of the many individuals who have worked tirelessly for the restoration and protection of the critical coastal resources of Barnstable and Dennis and drove for the completion of the Linked Watershed-Embayment Model to Determine the Critical Nitrogen Loading Threshold for the Barnstable Great Marshes - Bass Hole Estuarine System. Without these stewards and their efforts, this project would not have been possible.

First and foremost we would like to recognize and applaud the commitment shown by both the Town of Dennis and the Town of Barnstable in carrying forward with the Massachusetts Estuaries Project and the protection/restoration of all the estuaries of the Towns. Significant time and attention has been dedicated to this effort by Mr. David Larkowski, whose efforts were instrumental to completion of this report. Mr. Larkowski, Director of the Dennis Water District was critical to establishing the Town of Dennis Estuarine Water Quality Monitoring Program and dedicated much personal time and effort towards organizing and participating in sample collection to determine the nutrient related water quality for Bass Hole and other estuaries in the Town of Dennis. Without this baseline water quality data the present analysis would not have been possible.

The MEP Technical Team would also like to recognize and applaud the significant time and effort in data collection and discussion spent by members of the Town of Barnstable Water Quality Monitoring Program and SMAST "volunteers". The Monitoring of Barnstable Harbor was truly a coordinated joint effort between the Town staff, citizens and Coastal Systems Program scientific staff. These individuals gave of their time to collect nutrient related water quality from this system, without which the present analysis would not have been possible. Of particular note are Dale Saad for her energy and perseverance in the stewardship of one of the regions key coastal resources and John Jacobsen of the Town of Barnstable, both of whom have coordinated the Barnstable Town-wide Monitoring Program since its inception. Additionally, many in the Town of Barnstable helped in this effort, the Barnstable Department of Public Works and the Barnstable Shellfish Department, as well as the Barnstable Nutrient Management Committee. The technical team would like to specifically acknowledge the early efforts of Mark Els for helping to move this effort forward from its inception. In addition, the Three Bays Preservation Inc served as the major NGO promoting estuarine restoration and monitoring within the Town of Barnstable including the Barnstable Great Marshes.

In addition to local contributions, technical, policy and regulatory support has been freely and graciously provided by Paul Niedzwiecki and Tom Cambareri of the Cape Cod Commission. We are also thankful for the long hours in the field and laboratory spent by technical staff (Jen Benson, Michael Bartlett, Sara Sampieri and Dahlia Medieros), interns and students within the Coastal Systems Program at SMAST-UMD.

Support for this project was provided by the Town of Barnstable, Town of Dennis, Barnstable County and the MassDEP.

PROPER CITATION

Howes B., E. Eichner, S. Kelley, R. Samimy, J. S. Ramsey, D. Schlezinger, P. Detjens (2017). Massachusetts Estuaries Project Linked Watershed-Embayment Model to Determine the Critical Nitrogen Loading Threshold for the Barnstable Great Marshes - Bass Hole Estuarine System, Town of Barnstable and Dennis, Massachusetts, Massachusetts Department of Environmental Protection. Boston, MA.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

I. INTRODUCTION

The Barnstable Great Marsh estuary system (inclusive of Bass Hole) is located within the Towns of Barnstable, Yarmouth, and Dennis on Cape Cod Massachusetts. The system has a northern shore bounded by a narrow barrier beach (Sandy Neck) separating the Harbor from Cape Cod Bay, with which it exchanges tidal waters. The Barnstable Harbor Estuary is one of the largest embayments on Cape Cod and is comprised of large open water areas (namely Barnstable Harbor) as well as a large salt marsh system (Great Marshes) in the western portion of the overall embayment. Barnstable Harbor also supports small salt marsh dominated tributary sub-embayments such as Bass Hole-Chase Garden Creek near to the inlet to the system as well as the Millway which supports an active marina and is the discharge point for the Maraspin Creek salt marsh (Figure I-1). The watershed contributing nitrogen to the waters of the Barnstable Harbor Estuary contains portions of the Towns of Barnstable, Yarmouth, Sandwich, and Dennis. The sub-watershed to Bass Hole-Chase Garden Creek, which is a large subsystem on the eastern side of the Barnstable Great Marsh estuary system, is located in the Towns of Dennis and Yarmouth. Protection/Restoration of healthy or degraded habitats within the estuary will depend mainly upon the efforts of the Town of Barnstable and its citizens, however, depending on the level of nutrient management there may be the need for some coordination of efforts with the Towns of Dennis and Yarmouth.

The present configuration of the Barnstable Harbor embayment system results from a combination of glacially dominated geologic processes including the deposition of glacial outwash deposits and tidal flooding of drowned river valleys (Alder Creek, Bridge Creek, Boat Cove Creek, Marapsin Creek, Chase Garden Creek, Whites Brook) formed primarily by post-glacial rivers and enhancements to support human uses (e.g. tidal channel to Millway). The major drowned-river valley components are found in the Great Marshes and associated tributary salt marshes such as Bass Hole-Chase Garden Creek. Overall, the Barnstable Great Marsh estuary system is a composite or complex estuary comprised of the aforementioned drowned river valley sub-estuaries exchanging tidal waters with a large lagoonal estuary, Barnstable Harbor, which is flushed by Cape Cod Bay. The large open water basin that is regarded as Barnstable Harbor is oriented in an east-west manner with a central axis that runs parallel to the shore line and is bounded to the north by barrier beach and to the south by the uplands of Cape Cod. The lagoon represents more than 1/2 of the estuarine area and habitat. The lagoon was mainly formed by the depression created by the overlying Cape Cod Lobe of the continental glacier that formed Cape Cod and occupied Cape Cod Bay during the last glacial period. The wetland shield that generally surrounds the lagoon was enhanced by sediments deposited in a lake that occupied most of the southern portion of the current Cape Cod Bay during the retreat of the glacier. Following the retreat of the glaciers, sea level gradually rose and drove erosion and sediment transport along the shores of Cape Cod. Sediment from the eroding Cape Cod Bay-side uplands to the west of Barnstable supplied sand to the bay side beaches by long shore drift. The sand moving east formed a spit of land off of the Barnstable mainland and an open water lagoon with associated salt marsh formation along the edges of the lagoon. The spit (Sandy Neck) formed through steady longshore transport of sand to the west and provided protection from the waves of Cape Cod Bay thus forming the sheltered environment of Barnstable Harbor with its associated salt marshes. The Barnstable Great Marsh estuary system is a relatively "young" estuary and coastal feature that required significant post glaciation sea-level rise and the formation of the barrier beach, occurring on the order of 2500-4000 years b.p.

Figure I-1. Study region proximal to the Barnstable Harbor embayment system for the Massachusetts Estuaries Project nitrogen thresholds analysis. Tidal waters enter the system through one wide "inlet" to Cape Cod Bay. Freshwaters enter from the watershed primarily through 6 notable surface water discharges: Alder Creek, Bridge Creek, Boat Cove Creek, Maraspin Creek Chase Garden Creek and White Brook, as well as direct groundwater discharge. The main basin constitutes Barnstable Harbor.

Although erosional processes associated with post-glacial streams and rivers were fundamental to the formation of portions of this system, at present the streams are relatively small and discharge only a small portion of the aquifer recharge to the estuary. The biggest of the streams is the Boat Cove Creek which discharges to the innermost portion of the Great Marshes portion of the overall system. The other five small freshwater creeks discharge at various points along salt marsh that fringes the southern side of the harbor. Most freshwater from the watershed enters the embayment through direct groundwater seepage along the southern shore after passing through seeps along the edges of the system's wetland shield.

As is typical of many other Cape Cod embayments (Nauset System, Pleasant Bay, Wellfleet Harbor), Barnstable Harbor is separated from Cape Cod Bay by a barrier beach, which is heavily influenced by coastal storms. Within portions of Barnstable Harbor, mainly Boat Cove Creek and Maraspin Creek, the tide propagating through the wide opening of the Harbor is attenuated by culverts which reduce tidal flow into the interior portions of these sub-systems. Generally, all other areas of the Barnstable Great Marsh estuary system experience an undampened tide and are relatively well flushed.

The barrier beach (Sandy Neck) that protects Barnstable Harbor to the north is a very dynamic geomorphic feature, due to the strong influence of littoral transport processes. While the formation of the Barnstable Great Marsh estuary system was dependent upon coastal processes which formed the barrier beach to form the lagoon, the estuary continues to be affected by these same coastal processes as they alter both the length of the spit extending eastward towards the inlet and the location of the tidal flats at the inlet. The effect of these processes is no longer to significantly affect the geomorphology of the estuary and its basins, but to partially control the quality of the habitats within the estuary given how they may be influenced by activities in the contributing watershed. Changes in hydrodynamics wrought by inlet and shoal dynamics is a key factor in determining the effects of watershed nitrogen loading on estuarine health (see Sections V & VI). To the extent that the small inlets to the sub-components of Barnstable Harbor become restricted due to shifting shoals and tidal flushing is reduced, nitrogen loading impacts will be magnified over present conditions. This effect is moderated by the fact that large sections of the western and southern portions of the overall system are dominated by salt marsh.

Similar to the Wellfleet Harbor, Nauset Harbor and Pleasant Bay embayment systems, Barnstable Harbor is a shallow coastal estuary dominated by salt marsh and tidal flats, as well as being located within a watershed that includes lake deposits, beach/dune and marsh deposits as well as moraine deposits, generally laid down after the retreat of the Cape Cod Lobe of the Laurentide Ice sheet ~15,000 years ago (Figure I-2, Oldale, 1992). The upper portions of the watershed are generally composed of both younger and older outwash deposits. The outwash material is highly permeable and varies in composition from well sorted medium sands to course pebble sands and gravels. As such, direct rainwater run-off is typically rather low for these coastal systems and therefore, most freshwater inflow to these estuarine systems is via groundwater discharge or groundwater fed surface water flow (e.g. Alder Creek, Boat Cove Creek, Maraspin Creek). Barnstable Harbor acts as a large mixing zone for terrestrial freshwater inflow and saline tidal flow from Cape Cod Bay, however, the salinity characteristics of the embayment system are mainly dominated by that of Cape Cod Bay with the exception of the uppermost reaches of the salt marsh tidal creeks that receive fresh surface water discharges. Given the large tidal flows and volumetric exchange, there is presently only minor dilution of salinity throughout most of the estuary.

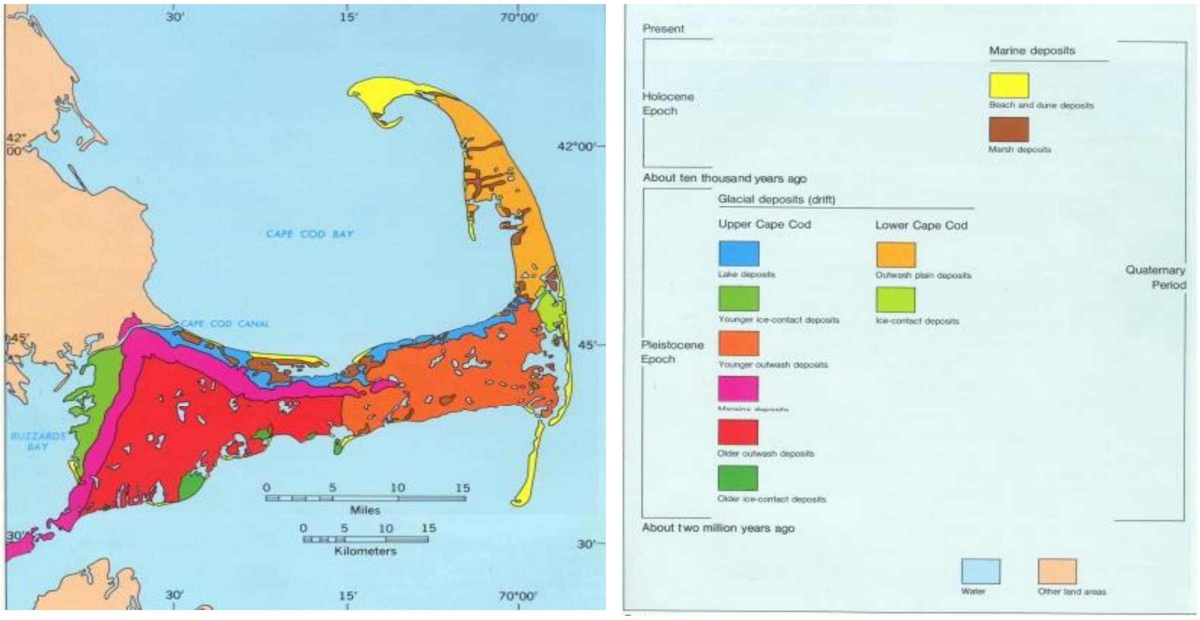


Figure I-2. Geologic map of Cape Cod (generalized from detailed mapping by K. F. Mather, R. P. Goldthwait, L. R. Theismeyer, J. H. Hartshorn, Carl Koteff, and R. N. Oldale).

Barnstable Harbor, along with its associated terminal sub-embayments which are dominated by salt marshes, constitutes an important component of the natural and cultural resources of Cape Cod and the Towns of Barnstable, Yarmouth, and Dennis. As such the Towns of Barnstable and Dennis, working with the Coastal Systems Program at the University of Massachusetts School for Marine Science and Technology, have undertaken comprehensive water quality monitoring of the Barnstable Great Marsh estuary system.

The primary ecological threat to Barnstable Harbor resources is degradation resulting from nutrient enrichment. Loading of the critical eutrophying nutrient, nitrogen, to the embayment waters has been greatly increased over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to this and other outer Cape embayment systems such as Pleasant Bay, Wellfleet Harbor and Nauset Harbor, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater. The Towns of Barnstable, Yarmouth, and Dennis have been steadily growing over the past two to three decades and Barnstable does have limited centralized wastewater treatment to process the increasing levels of nutrients resulting from the increased development. As existing and probable increasing levels of nutrients impact the coastal embayments of Barnstable Harbor, the potential for water quality degradation will increase, with further harm to invaluable environmental resources of the Towns.

The large shoreline and numerous streams greatly increases the potential for direct discharges from homes situated on the shore and decreases the travel time of groundwater from areas removed from the shoreline and close to the watershed boundary. The nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine shoreline they are popular regions for boating, recreation, and land development; as semi-enclosed bodies of water, they may not be readily flushed of the pollutants that they receive due to the proximity and density of development near and along their shores. In particular, the more enclosed portions of the basin (like the Millway) and portions furthest from the inlet are at risk of eutrophication from high nitrogen loads entering via direct groundwater seepage in addition to surface water inflows from adjacent sub-watersheds.

Given the value of the resource and concern over the problems associated with nutrient over-enrichment, in 1978 the Massachusetts Secretary of Environmental Affairs designated part of Barnstable Harbor as an Area of Critical Environmental Concern (ACEC). The ACEC boundary generally follows the 100-year floodplain elevation on the landward side and mean low water on the seaward side of the Harbor. Almost half of the land within the ACEC boundary is covered by salt marsh habitat, and over 75% of the area is located within the 100-year floodplain. The majority of the 4,335 acres of protected open space in the ACEC are owned by the Town of Barnstable - located on Sandy Neck and in the adjacent expanse of salt marsh, locally known as the Great Marsh. Approximately 90% of the ACEC exists within the Town of Barnstable and ~10% exists in the Town of Sandwich. There are also several private in-holdings, including two small (< 20 acre) tracts owned by The Nature Conservancy and cottages at the end of the Neck, which are leased by the town to private individuals. The purpose of the ACEC program is to preserve, restore, and enhance critical environmental resource areas in the state. The designation is intended to encourage communities to steward the area's natural resources, but in practical terms it provides little regulatory oversight. It has therefore been necessary for the Town of Barnstable and Sandwich to take the initiative to provide such oversight and clarity through amendments and revisions to town Environmental By-Laws and take necessary steps to protect the Barnstable Harbor resource consistent with the ACEC designation.

As the primary stakeholder to the Barnstable Harbor embayment system, the Town of Barnstable was among the first communities to become concerned over perceived degradation of embayment health. The Town of Barnstable (via the Town of Barnstable Department of Public Works) has long recognized the potential threat of nutrient over-enrichment of the Town's coastal embayments. As such, a comprehensive water quality monitoring program was developed to establish the current water quality conditions in the harbor and monitor for gradual changes in water quality over time. The common focus of the water quality monitoring efforts undertaken by the Town of Barnstable has been to gather site-specific data on the current nitrogen related water quality throughout the Barnstable Great Marsh estuary system to ultimately determine the relationship between observed water quality and watershed nitrogen loads. This multi-year effort has provided the baseline information required for determining the link between upland loading, tidal flushing, and estuarine water quality. The water quality data set developed by the Town of Barnstable Water Quality Monitoring Program over the past 5+ years forms a baseline from which to gauge long-term changes as watershed nitrogen management moves forward. The quality of these data allowed the MEP to prioritize the Barnstable Great Marsh estuary system for this next step in the protection and management of the harbor.

The MEP effort builds upon the efforts of the water quality monitoring programs, and previous hydrodynamic and water quality analyses, and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Barnstable Great Marsh estuary system, including all sub-embayments such as the Millway and Bass Hole in addition to specific consideration of the Great Marshes portion of the overall embayment.

The critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to undertake wastewater master planning and nitrogen management alternatives development which may be currently needed by the Town of Barnstable, Yarmouth, and Dennis. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff, committees, and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns to develop and evaluate the most cost effective nitrogen management alternatives to protect/restore this valuable coastal resource which is currently being degraded by nitrogen overloading.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

Coastal embayments throughout the Commonwealth of Massachusetts (and along the U.S. eastern seaboard) are becoming nutrient enriched. The nutrients are primarily related to changes in watershed land-use associated with increasing population within the coastal zone over the past half century. Many of Massachusetts' embayments have nutrient levels that are approaching or are currently over this assimilative capacity, which begins to cause declines in their ecological health. The result is the loss of fisheries habitat, eelgrass beds, and a general disruption of benthic communities and the food chain which they support. At higher levels, nitrogen loading from surrounding watersheds causes aesthetic degradation and inhibits even recreational uses of coastal waters. In addition to nutrient related ecological declines, an increasing number of embayments are being closed to swimming, shellfishing and other activities as a result of bacterial contamination. While bacterial contamination does not generally degrade the habitat, it restricts human uses. However like nutrients, bacterial contamination is frequently related to changes in land-use as watersheds become more developed. The regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to the culture, economy, and tax base of Massachusetts's coastal communities. This is particularly the case across Cape Cod and in the Town of Barnstable considering the significant embayments of Barnstable Harbor, Three Bays, Lewis Bay and Centerville Harbor.

The primary nutrient causing the increasing impairment of the Commonwealth's coastal embayments is nitrogen and the primary sources of this nitrogen are wastewater disposal, fertilizers, and changes in the freshwater hydrology associated with development. At present there is a critical need for state-of-the-art approaches for evaluating and restoring nitrogen sensitive and impaired embayments. Within southeastern Massachusetts alone, almost all of the municipalities (as is the case with the Town of Wellfleet) are grappling with Comprehensive Wastewater Planning and/or environmental management issues related to the declining health of their estuaries.

Municipalities are seeking guidance on the assessment of nitrogen sensitive embayments, as well as available options for meeting nitrogen goals and approaches for restoring impaired systems. Many of the communities have encountered problems with "first generation" watershed based approaches, which do not incorporate estuarine processes. The appropriate method must be quantitative and directly link watershed and embayment nitrogen conditions. This "Linked" Modeling approach must also be readily calibrated, validated, and implemented to support planning. Although it may be technically complex to implement, results must be understandable to the regulatory community, town officials, and the general public.

The Massachusetts Estuaries Project represents the next generation of watershed based nitrogen management approaches. The Massachusetts Department of Environmental Protection (MassDEP), the University of Massachusetts – Dartmouth School of Marine Science and Technology (SMAST), and others including the Cape Cod Commission (CCC) have undertaken the task of providing a quantitative tool for watershed-embayment management for communities throughout Southeastern Massachusetts. The MEP approach was selected after extensive review by the MassDEP and USEPA and associated scientists and engineers. It has subsequently been applied to more than 60 estuaries and reviewed by other state agencies, municipalities, non-profit environmental organizations, engineering firms, scientists and private citizens. Over the course of the extensive reviews, the MEP approach has proven to be robust and capable of yielding quantitative results to support management of a wide variety of estuaries.

The Massachusetts Estuary Project is founded upon science-based management. The Project is using a consistent, state-of-the-art approach throughout the region's coastal waters and providing technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the Massachusetts Estuaries Project is to provide the DEP and municipalities with technical guidance to support policies on nitrogen loading to embayments. In addition, the technical reports prepared for each embayment system in the MEP study region is serving as the basis for the development of Total Maximum Daily Loads (TMDLs). Development of TMDLs is required pursuant to Section 303(d) of the Federal Clean Water Act. TMDLs must identify sources of the pollutant of concern (in this case nitrogen) from both point and non-point sources, the allowable load to meet the state water quality standards and then allocate that load to all sources taking into consideration a margin of safety, seasonal variations, and several other factors. In addition, each TMDL must contain an outline of an implementation plan. For this project, the DEP recognizes that there are likely to be multiple ways to achieve the desired goals, some of which are more cost effective than others and therefore, it is extremely important for each Town to further evaluate potential options suitable to their community. As such, DEP will likely be recommending that specific activities and timelines be further evaluated and developed by the Towns (sometimes jointly) through the Comprehensive Wastewater Management Planning process.

The MEP nitrogen threshold analysis includes site-specific habitat assessments and watershed/embayment modeling approaches to develop and assess various nitrogen management alternatives for meeting selected nitrogen goals supportive of restoration/protection of embayment health.

The major MEP nitrogen management goals are to:

- provide technical analysis and supporting documentation to Towns as a basis for sound nutrient management decision making towards embayment restoration
- develop a coastal TMDL working group for coordination and rapid transfer of results,
- determine the nutrient sensitivity of each of 70 embayments in Southeastern MA
- provide necessary data collection and analysis required for quantitative modeling,
- conduct quantitative TMDL analysis, outreach, and planning,
- keep each embayment's model "alive" to address future municipal needs.

The core of the Massachusetts Estuaries Project analytical method is the Linked Watershed-Embayment Management Modeling Approach (Figure I-3). This approach represents the "next generation" of nitrogen management strategies. It fully links watershed inputs with embayment circulation and nitrogen characteristics. The Linked Model builds on and refines well accepted basic watershed nitrogen loading approaches such as those used in the Buzzards Bay Project, the CCC models, and other relevant models. However, the Linked Model differs from other nitrogen management models in that it:

- requires site specific measurements within each watershed and embayment;

- uses realistic “best-estimates” of nitrogen loads from each land-use within the watershed (as opposed to loads with built-in “safety factors” like Title 5 design loads);
- spatially distributes the watershed nitrogen loading to the embayment;
- accounts for watershed nitrogen attenuation during transport to the embayment;
- includes a 2D or 3D embayment circulation model depending on embayment structure;
- accounts for basin structure, tidal variations, and dispersion within the embayment;
- includes nitrogen regenerated from sediments within the embayment;
- is validated by both independent hydrodynamic, nitrogen and salinity concentrations, and ecological data;
- is calibrated and validated with field data prior to generation of “what if” scenarios.

The Linked Model has been applied for watershed nitrogen management of more than 60 embayments throughout southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach’s greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing “what if” scenarios for evaluating watershed nitrogen management options. The MEP Technical Team, through SMAST-UMD, has conducted more than 200 scenarios to date.

The Linked Watershed-Embayment Model when properly parameterized, calibrated and validated for a given embayment becomes a nitrogen management planning tool, which fully supports TMDL analysis. The Model facilitates the evaluation of nitrogen management alternatives relative to meeting water quality targets within a specific embayment. The Linked Watershed-Embayment Model also enables Towns to evaluate improvements in water quality relative to the associated cost. In addition, once a model is fully functional it can be “kept alive” and updated for continuing changes in land-use or embayment characteristics (at minimal cost). In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries.

Linked Watershed-Embayment Model Overview: The Model provides a quantitative approach for determining an embayment’s: (1) nitrogen sensitivity, (2) nitrogen threshold loading levels (TMDL) and (3) response to changes in loading rate. The approach is both calibrated and fully field validated and unlike many approaches, accounts for nutrient sources, attenuation, and recycling and variations in tidal hydrodynamics (Figure I-3). This methodology integrates a variety of collected field data and models, specifically:

- Water column Monitoring - multi-year embayment nutrient sampling
- Hydrodynamics -
  - - embayment bathymetry
  - - site specific tidal record
  - - current records (in complex systems only)
  - - hydrodynamic model
- Watershed Nitrogen Loading
  - - watershed delineation
  - - stream flow (Q) and nitrogen load
  - - land-use analysis (GIS)
  - - watershed N model built from individual parcels
- Embayment TMDL - Synthesis
  - - linked Watershed-Embayment N Model
  - - salinity surveys (for linked model validation)

- rate of N recycling within embayment
- continuous dissolved oxygen record
- Macrophyte survey
- Infaunal survey

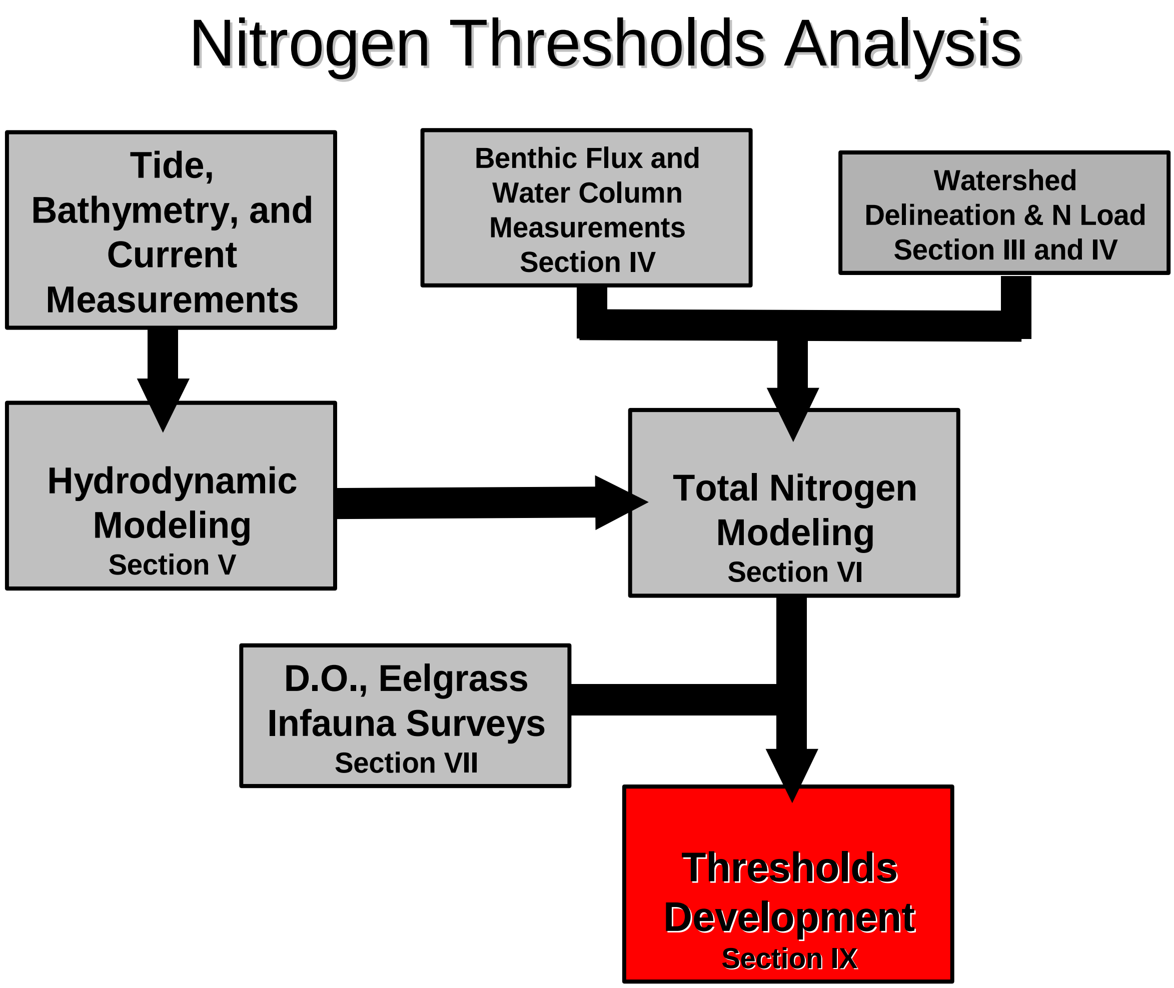


Figure I-3. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach

As management alternatives are being developed and evaluated, it is important to note that nitrogen loading and tidal exchange within each sub-embayment is the primary factor controlling habitat health in that sub-basin. The quality of the inflowing waters from Barnstable Harbor to tributary sub-embayments is the other, although a slightly less critical controlling factor given the connectivity to low nutrient water from Cape Cod Bay. In addition the nitrogen loading to each sub-embayment affects the health of the receiving main basin of the System. Much of the nitrogen entering the lagoonal component first passes through a sub-embayment. The result is that the restoration of potentially nitrogen impaired sub-embayments to the Barnstable Great Marsh estuary system require both "local" or contributing area specific nitrogen management, as well as management of nitrogen levels within the watershed of the larger "regional" main basin.

I.2 NITROGEN LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In sandy glacial outwash aquifers, such as in the watershed to the Barnstable Harbor embayment system, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer mineral (Weiskel and Howes 1992). Since even Cape Cod “rivers” are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In contrast, nitrogen, primarily as plant-available nitrate, is readily transported through oxygenated groundwater systems on Cape Cod (DeSimone and Howes 1998, Weiskel and Howes 1992, Smith *et al.* 1991). The result is that terrestrial inputs to coastal waters tend to be higher in plant available nitrogen than phosphorus (relative to plant growth requirements). However, coastal estuaries tend to have algal growth limited by nitrogen availability, due to their flooding with low nitrogen coastal waters (Ryther and Dunstan 1971). Tidal reaches within Barnstable Great Marsh estuary system follow this general pattern, where the primary nutrient of eutrophication in these systems is nitrogen.

Nutrient related water quality decline represents one of the most serious threats to the ecological health of the nearshore coastal waters. Coastal embayments, because of their enclosed basins, shallow waters and large shoreline area, are generally the first indicators of nutrient pollution from terrestrial sources. By nature, these systems are highly productive environments, but nutrient over-enrichment of these systems worldwide is resulting in the loss of their aesthetic, economic and commercially valuable attributes.

Each embayment system maintains a capacity to assimilate watershed nitrogen inputs without degradation. However, as loading increases a point is reached at which the capacity (termed assimilative capacity) is exceeded and nutrient related water quality degradation occurs. As nearshore coastal salt ponds, marshes and embayments are the primary recipients of nutrients carried via surface and groundwater transport from terrestrial sources, it is clear that activities within the watershed, often miles from the water body itself, can have chronic and long lasting impacts on these fragile coastal environments.

Protection and restoration of coastal embayments from nitrogen overloading has resulted in a focus on determining the assimilative capacity of these aquatic systems for nitrogen. While this effort is ongoing (e.g. USEPA TMDL studies), southeastern Massachusetts has been the site of intensive efforts in this area (Eichner et al., 1998, Costa et al., 1992 and in press, Ramsey et al., 1995, Howes and Taylor, 1990, the Falmouth Coastal Overlay Bylaw). While each approach may be different, they all focus on changes in nitrogen loading from watershed to embayment, and aim at projecting the level of increase in nitrogen concentration within the receiving waters. Each approach depends upon estimates of circulation within the embayment; however, few directly link the watershed and hydrodynamic models, and virtually none include internal recycling of nitrogen (as was done in the present effort). However, determination of the “allowable N concentration increase” or “threshold nitrogen concentration” used in previous studies had a significant uncertainty due to the need for direct linkage of watershed and embayment models and site-specific data. In the present effort we have integrated site-specific data on nitrogen levels and the gradient in N concentration throughout the Barnstable Great Marsh estuary system monitored by the Town of Barnstable and the Town of Dennis Water Quality Monitoring Programs with site-specific habitat quality data (D.O., eelgrass, phytoplankton blooms, benthic animals) to “tune” general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

In general, nutrient over-fertilization is termed “eutrophication” and when the nutrient loading is primarily from human activities, “cultural eutrophication”. Although the influence of human-induced changes has increased nitrogen loading to the Barnstable Great Marsh estuary system and potentially contributed to the degradation in ecological health, it is sometimes possible that eutrophication within portions of the Barnstable Great Marsh estuary system could potentially occur without anthropogenic influence and this must always be considered in the nutrient threshold analysis. While this finding would not change the need for protection or restoration, it would change the approach and potential targets for management. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a “pristine” system.

I.3 WATER QUALITY MODELING

Evaluation of upland nitrogen loading provides important “boundary conditions” for water quality modeling of the Barnstable Harbor-Great Marshes system; however, a thorough understanding of estuarine circulation is required to accurately determine nitrogen concentrations within the system. Therefore, water quality modeling of tidally influenced estuaries must include a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Numerical models provide a cost-effective method for evaluating tidal hydrodynamics since they require limited data collection and may be utilized to numerically assess a range of management alternatives. Once the hydrodynamics of an estuary system are understood, computations regarding the related coastal processes become relatively straightforward extensions to the hydrodynamic modeling. The spread of pollutants may be analyzed from tidal current information developed by the numerical models.

The MEP water quality evaluation examined the potential impacts of nitrogen loading into the Barnstable Great Marsh estuary system and all of its component sub-embayments (e.g. Bass Hole). A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for the system. Once the hydrodynamic properties of the estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis defined by MEP watershed delineations based on USGS recharge area outputs from the regional West Cape model. Almost all nitrogen entering the Barnstable Great Marsh estuary system is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Cape Cod Bay source waters and throughout the Barnstable Great Marsh estuary system was taken from the water quality monitoring program run by the Town of Barnstable (associated with the Coastal Systems Program at SMAST). Measurements of current salinity and nitrogen and salinity distributions throughout estuarine waters of the system were used to calibrate and validate the water quality model (under existing loading conditions).

I.4 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Barnstable Great Marsh estuary system (inclusive of Bass Hole) for the Towns of Barnstable and Dennis. A review of existing studies related to habitat health or nutrient related water quality is provided in Section II with a more detailed review of prior hydrodynamic investigations in Section V. The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described, along with streamflow measurements and loads. Since nitrogen recycling associated with the bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component was also summarized in Section IV. Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from town-supplied data and offshore water column nitrogen values were derived from an analysis of monitoring station data for an offshore station proximal to the “inlet” of the Harbor (Section IV and VI). Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data (conducted by the municipality) as discussed in Section VI. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling, as well as an analysis of how the measured nitrogen levels correlate to observed estuarine water quality are described in Section VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of each embayment was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of each embayment in Section VIII.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

Nutrient additions to aquatic systems cause shifts in a series of biological processes that can result in impaired nutrient-related habitat quality. Effects include excessive plankton and macrophyte growth, which in turn lead to reduced water clarity, organic matter enrichment of waters and sediments with concomitant increased rates of oxygen consumption and periodic depletion of dissolved oxygen, especially in bottom waters, and the limitation of the growth of desirable species such as eelgrass. Even without changes to water clarity and bottom water dissolved oxygen, the increased organic matter deposition to the sediments generally results in a decline in habitat quality for benthic animal communities (e.g. "infauna", animals living in the sediments). This habitat change causes a shift in infaunal communities from high diversity, deep burrowing forms (which include economically important species), to low diversity shallow dwelling organisms. This shift alone causes significant degradation of the resource and a loss of productivity important for local shell fisherman and sport and offshore fin fisheries, which are dependent upon these highly productive estuarine systems as a habitat and food resource during migration or during different phases of their life cycles. In addition, the diverse avian fauna which feed upon infauna or dependent fish communities are also affected and their numbers and diversity decline. This overall nutrient driven process is generally termed “eutrophication” and in embayment systems, unlike in shallow lakes and ponds, it is not necessarily a part of the natural evolution of a system.

In most marine and estuarine systems, such as the Barnstable Great Marsh estuary system, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if the level of nitrogen enrichment is controlled, then eutrophication is controlled. As a result, there has been significant effort to develop tools for predicting how modification of watershed nitrogen loads and changes in tidal flushing quantitatively cause changes in the concentrations of water column nitrogen in the receiving estuary. Further development of these approaches generated specific guidelines as to what is to be considered acceptable water column nitrogen concentrations to achieve desired water quality goals (e.g., see Cape Cod Commission 1991, 1998; Howes et al. 2002).

These previous tools for predicting loads and concentrations tend to be generic in nature, and overlook some of the specifics for any given water body. In contrast, some approaches can be tailored for each individual estuary of interest, but require large amounts of site-specific information and therefore are not generally applied. The present Massachusetts Estuaries Project (MEP) effort uses one such site-specific approach, but includes a tailored and targeted data collection approach that is “parsimonious” in its level of effort (MEP Peer Review, 2011). The assessment focuses on linking water quality model predictions, based upon watershed nitrogen loading and embayment recycling and system hydrodynamics, to actual measured values for specific nutrient species within individual estuaries. The linked watershed-embayment model is built using embayment- and watershed-specific measurements, thus enabling calibration of the prediction process for the specific conditions in each of the coastal embayments of southeastern Massachusetts, including the Barnstable Great Marsh estuary system. As the MEP approach requires substantial amounts of site-specific data collection, part of the program is to review previous data collection and modeling efforts. These reviews are both for purposes of “data mining” and to gather additional information on an estuary’s habitat quality and unique features.

Few studies relating to nitrogen loading, hydrodynamics and habitat health have been conducted within the Barnstable Great Marsh estuary system (inclusive of Bass Hole) over the past two decades to help inform the MEP process. Directly supporting the present Massachusetts

Estuaries Project effort to develop a nitrogen threshold for the Barnstable Harbor estuary was the Barnstable Harbor Nutrient Related Water Quality Monitoring Program. Water quality monitoring stations in Barnstable Harbor were sampled through collaborative effort between the Town of Barnstable and scientists from the Coastal Systems Program at UMD-SMAST. Water quality monitoring stations in Bass Hole were sampled by Town of Dennis staff with assays also conducted by the Coastal Systems Program Laboratory. This previous work provided quantitative information on water column parameters over multiple summers (including nitrogen) and was used in the calibration and verification of the MEP water quality model for this estuary. Available previous studies are summarized below.

Barnstable Harbor Nutrient Related Water Quality Monitoring: The MEP analysis requires high quality water quality data in order to complete its assessment and modeling approach. Baseline water quality monitoring was undertaken in the Barnstable Great Marsh estuary system as well as Bass Hole. The Town of Barnstable Water Quality Monitoring Program collected data on nutrient-related water quality throughout the estuarine systems of the Town of Barnstable (specifically the Harbor and Great Marsh) whereas data from Bass Hole was generated through the Town of Dennis Water Quality Monitoring Program. Both programs were developed with technical support from the SMAST Coastal Systems Program and all water quality samples were assayed at the SMAST Coastal Systems Program Laboratory. The Town of Barnstable and Dennis Water Quality Monitoring Programs have collected the principal baseline water quality data necessary for ecological management of each portion of the overall estuary. Both the monitoring programs (Barnstable and Dennis) are citizen-based water quality monitoring programs run by the Towns. In Barnstable the program is run through the Department of Public Works (D. Saad, Town of Barnstable Senior Project Manager). In Dennis, the water quality monitoring program was initiated through the Dennis Water District (D. Larkowski) and then turned over to the Town of Dennis (Water Quality Advisory Committee). Both town programs were provided with technical and analytical assistance from the Coastal Systems Program at SMAST-UMD. During the period the town monitoring programs were actively collecting MEP water samples, the monitoring program approaches had a USEPA and MassDEP approvals (MEP Quality Assurance Project Plan, 2003, Dennis SAP, 2005) and the SMAST Coastal Systems Program Laboratory had USEPA approval for all assay procedures.

During the course of the multi-year sampling program, water samples were collected from each station in each portion of the overall system during 6 sample rounds from June through mid-September starting in 2005 and continuing to present. Sampling was undertaken in this manner to target what is typically the period of poorest nutrient related water quality that is the focus of managing these systems. Marine stations were sampled at approximately two-week intervals during the falling tide (targeting the 2 hours before and after mid-ebb) during the early morning hours (6 to 9 A.M.). In general terms, the Town of Barnstable collected samples from 6 "open" water stations in the Harbor, CSP-SMAST scientific staff collected samples from 5 salt marsh stations and Town of Dennis staff collected samples from 7 stations distributed throughout the Bass Hole salt marsh, which discharges to the Barnstable Great Marsh estuary system.

The common focus of the Town of Barnstable and Dennis Water Quality Monitoring Program efforts has been to gather site-specific data on the current nitrogen-related water quality throughout the Harbor and associated salt marshes to support evaluations of observed water quality and habitat health. The Monitoring Programs generated a data set that elucidated the long-term water quality of the overall system (Figure II-1) and served as the basis for calibration and validation of the MEP water quality model. The monitoring was undertaken as a

Figure II-1. Town of Barnstable and Dennis Water Quality Monitoring Program for Barnstable Harbor (stations "BM" and "GM") and Bass Hole (stations "BSH") respectively. Estuarine water quality monitoring stations sampled by Town of Barnstable and Dennis staff and volunteers and analyzed by SMAST staff during summers 2005, 2006, 2007.

collaborative effort with the Towns coordinating the field effort and chemical assays being completed by the SMAST Coastal Systems Analytical Facility. The Coastal Systems Analytical Facility is located in the School for Marine Science and Technology UMASS-Dartmouth, 706 S. Rodney French Blvd, New Bedford, MA, and the laboratory Points of Contact are Sara Sampieri 508-910-6325 (ssampieri@umassd.edu) or Mike Bartlett (mbartlett@umassd.edu). Use of the SMAST Analytical Facility ensured sufficient sensitivity and accuracy of the analytical protocols and that proper QA/QC procedures were followed to allow incorporation of the data into the MEP analysis. Since the MEP approach relies on site-specific data, the baseline water quality data collected by the towns were a prerequisite to complete the MEP assessment of Barnstable Harbor. Implementation of the MEP's Linked Watershed-Embayment Approach necessarily incorporates the quantitative water column nitrogen data (2005-2015) gathered by the Monitoring Program and watershed and embayment data collected by MEP staff.

Since the results of the long term Water Quality Monitoring Program (2005-2015) and the above studies indicate that portions of the Barnstable Great Marsh estuary system could be threatened by the combination of land-derived nitrogen inputs and intermittent restriction of tidal exchange, the Towns of Barnstable, Dennis, and Yarmouth undertook participation in the Massachusetts Estuaries Project to complete ecological assessment and water quality modeling for the development of nutrient thresholds for protection of the Barnstable Harbor-Great Marshes system.

Initial Study of the Hydrodynamics and Flushing of Barnstable Harbor (ACRE, 1999): This study was completed in 1999 to evaluate the hydrodynamic characteristics of this system for purposes of calculating system flushing rates for the entire system as well as local residence times for selected sub-embayments. The study was performed by Applied Coastal Research and Engineering, Inc. (ACRE) for the Town of Barnstable, Massachusetts under a grant from the Cape Cod Commission. The purpose of the study was to evaluate flushing characteristics for selected areas to guide Town planning efforts within the system's watershed and develop a first order review of sensitivity of the system to watershed nitrogen loading (CCC, 2002).

To develop the flushing characteristics, ACRE (a MEP Technical Team member) developed a calibrated, two-dimensional numerical model of the estuary. This model relied upon data collected by Dutch researchers during an earlier study of the system in 1991-1993 (Van der Molen, 1997), along with tidal elevations collected by ACRE at six locations within the system. This study measured system geometry (marsh elevations, bathymetry) as well as tidal elevations. All collected data were used to generate a computer model that represented key features of the estuary. The model was calibrated against the tidal measurements to assure the essential hydrodynamics were being simulated correctly. Once calibrated, the model was used to calculate the tidal prism (water volume transported into the system during a single flood tide) and system and local residence times of water within various portions of the system.

Seasonal Changes in the Community Structure of the Macrobenthos Inhabiting the Intertidal Sand and Mud Flats of Barnstable Harbor, Massachusetts (R. Whitlatch, 1977): The purpose of this study was to describe the general macrobenthic community structure of a portion of the intertidal sand and mud flats of Barnstable Harbor, Massachusetts, and also to examine in greater detail seasonal changes in patterns of community structure and species composition at one of the sedimentary environments. The study was presented to provide a context for a more detailed examination of methods of resource allocation in deposit-feeding faunal assemblages and micro-dispersion patterns of surface-feeding polychaete annelids. Sixteen preliminary stations were sampled in the western portion of the intertidal sand and mud flats of Barnstable Harbor in June-July 1974. Faunal data collected from these stations were used to help in delimiting the general characteristics of the various macrofaunal assemblages of this region. One station was chosen from the initial sampling program for more detailed long-term sampling. Samples were collected at low tide with a small hand held core tube 12.5 cm in diameter. The core tube was gently pushed into the substrate to a depth of 17-20 cm and removed with a shovel. The location of sampling sites at each station was determined by random sampling coordinates. Samples were brought to a laboratory and gently washed through a 250 mesh screen. The residues were preserved in 70% alcohol, sorted under a dissecting microscope and the organisms encountered were identified to species. Ultimately, 64 core samples were obtained from the sampling of 16 stations in Barnstable Harbor. Of the 47 species found in the survey, the majority were polychaetes (29 species), crustaceans (9 species) and molluscs (6 species). This serves as a potentially meaningful point of comparison to the MEP infaunal analysis in order to determine how the benthic community has changed from 1974 to present. Among other findings, the study concluded that the majority of macrobenthos at Barnstable Harbor were deposit-feeders which comprised more than 90% of all organisms sampled. The deposit-feeders normally dominate mud and muddy-sand sediments. Suspension-feeders were most abundant in fine sands. The relationship of sedimentary parameters affecting the distribution of both trophic groups proposed by previous researchers (e.g. H. Sanders) is generally supported. While no significant changes were evident in species diversity, evenness, or species number throughout a 19 month sampling period, classification analysis delimited seasonal clustering of both samples and species groupings. These patterns were repeatable over a two year period suggesting that dynamic "equilibrium", no successional change was occurring in Barnstable Harbor at the time of the study.

Regulatory Assessments of Barnstable Harbor Resources - In addition to locally generated studies, the Barnstable Great Marsh estuary system is part of the Commonwealth's environmental surveys to support regulatory needs. The Barnstable Great Marsh estuary system contains a variety of natural resources of value to the citizens of Barnstable, Yarmouth, and Dennis as well as to the Commonwealth. As such, over the years surveys have been conducted to support protection and management of these resources. The MEP gathers the available information on these resources as part of its assessment, and presents some of them here for reference by those providing stewardship for this estuary and some in Section 7 to support the nitrogen thresholds analysis. For the Barnstable Great Marsh estuary system, available agency surveys include:

- Designated Shellfish Growing Area – MassDMF (Figure II-2a,b,c)
- Shellfish Suitability Areas – MassDMF (Figure II-3)
- Area of Critical Environmental Concern - ACEC (Figure II-4)
- Estimated Habitats for Rare Wildlife and State Protected Rare Species – NHESP (Figure II-5)
- Mouth of Coastal Rivers – MassDEP Wetlands Program (Figure II-6a,b,c,d,e,f,g,h)

The MEP effort builds upon earlier watershed delineations (Section III) and land-use analyses and water-use data (Section IV.1), historical eelgrass surveys (Section VII) and water quality surveys discussed above. This information is integrated with MEP higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Barnstable Harbor Estuary. The MEP has incorporated appropriate and available data from pertinent previous studies and available Town databases to enhance the determination of nitrogen thresholds for the Barnstable Great Marsh estuary system and to reduce costs to the Town of Barnstable and Dennis.

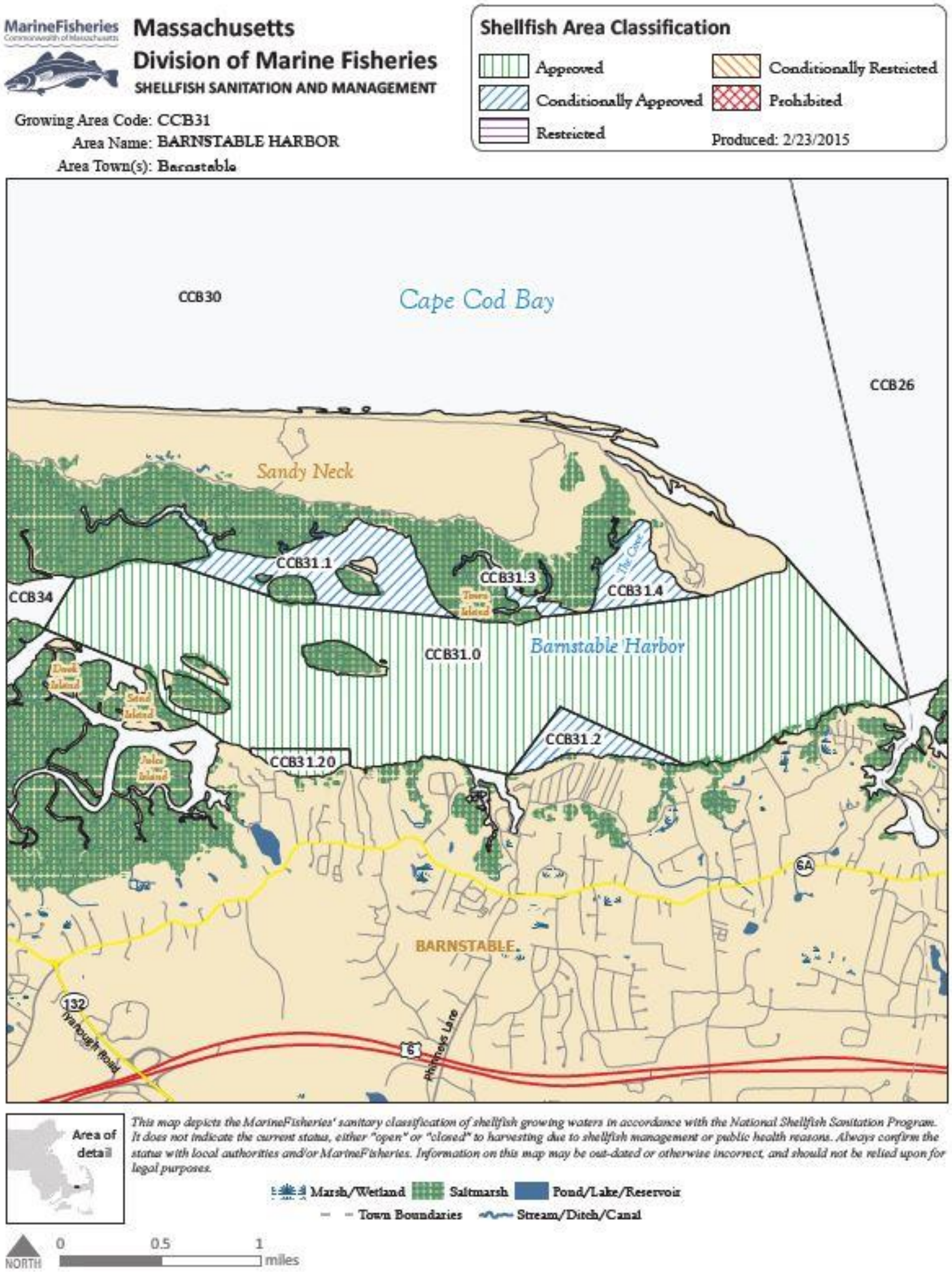


Figure II-2a. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas. However, areas dominated by wetlands with persistent fecal coliform levels >14 cfu per 100 mL may be prohibited to shellfishing until the cause of the contamination (frequently wildlife and birds) is documented.

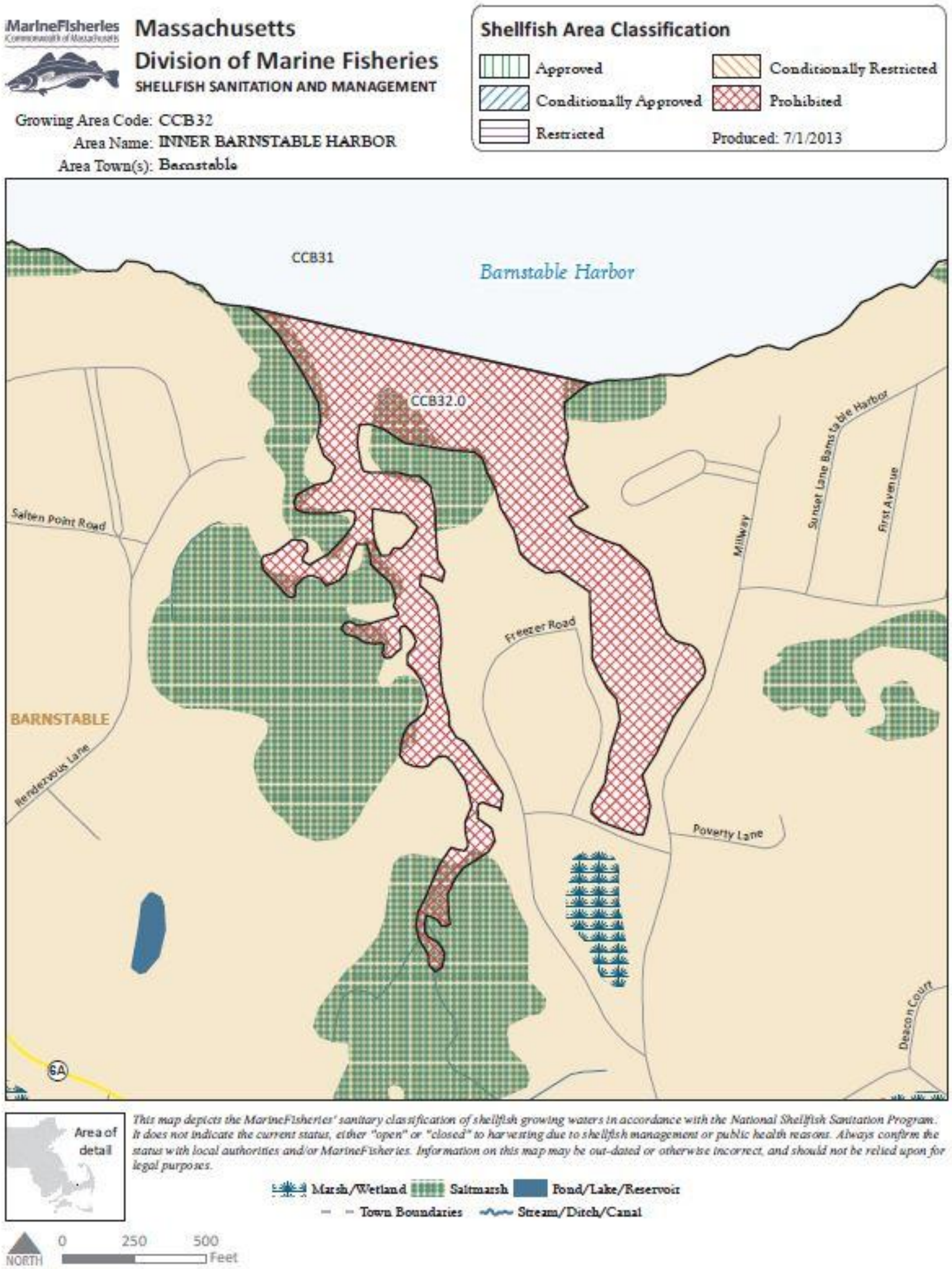


Figure II-2b. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas. However, areas dominated by wetlands with persistent fecal coliform levels >14 cfu per 100 mL may be prohibited to shellfishing until the cause of the contamination (frequently wildlife and birds) is documented.

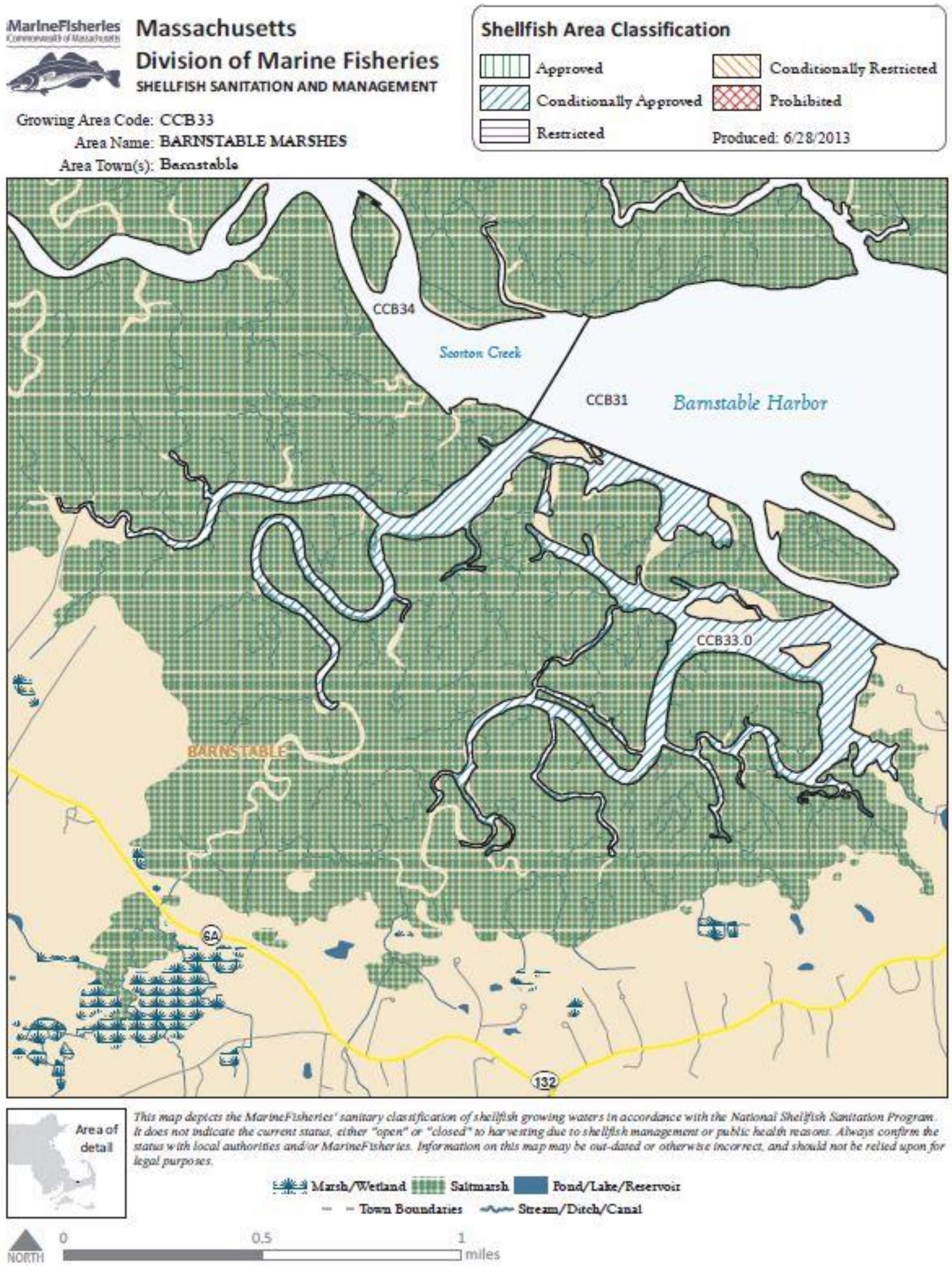


Figure II-2c. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas. However, areas dominated by wetlands with persistent fecal coliform levels >14 cfu per 100 mL may be prohibited to shellfishing until the cause of the contamination (frequently wildlife and birds) is documented.

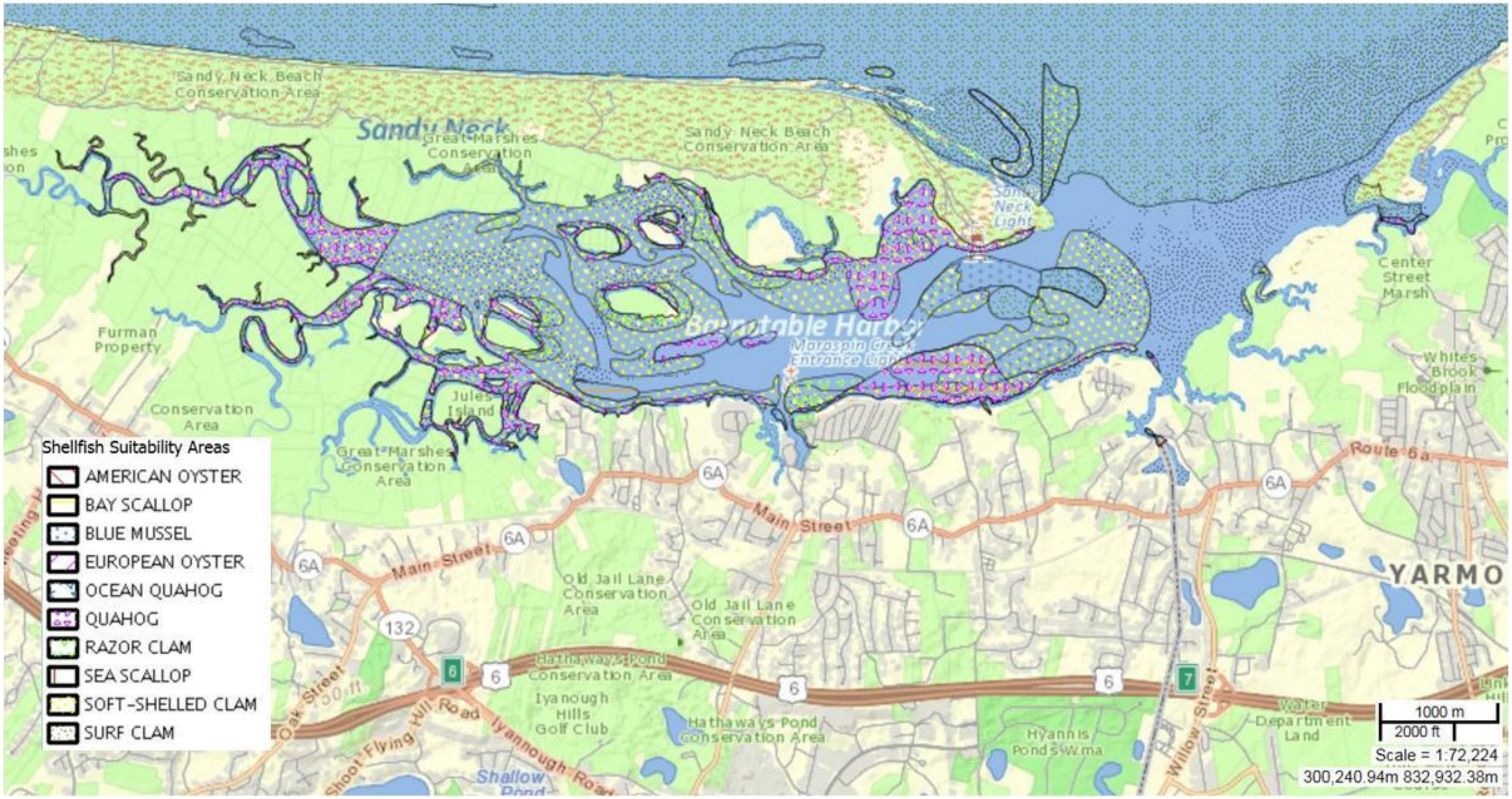


Figure II-3. Location of shellfish suitability areas within the Barnstable Harbor Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean "presence". The denoted shellfish areas are generally associated with the lower tidal reaches of the major tidal creeks, which support sediments comprised of medium to fine sands.

Figure II-4. Areas of Critical Environmental Concern (ACEC) within the Barnstable Harbor Estuary extending the Howland Lane / Route 6A interconnection to the Scorton Creek system.

Figure II-5. Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Barnstable Harbor Estuary as determined by - NHESP.

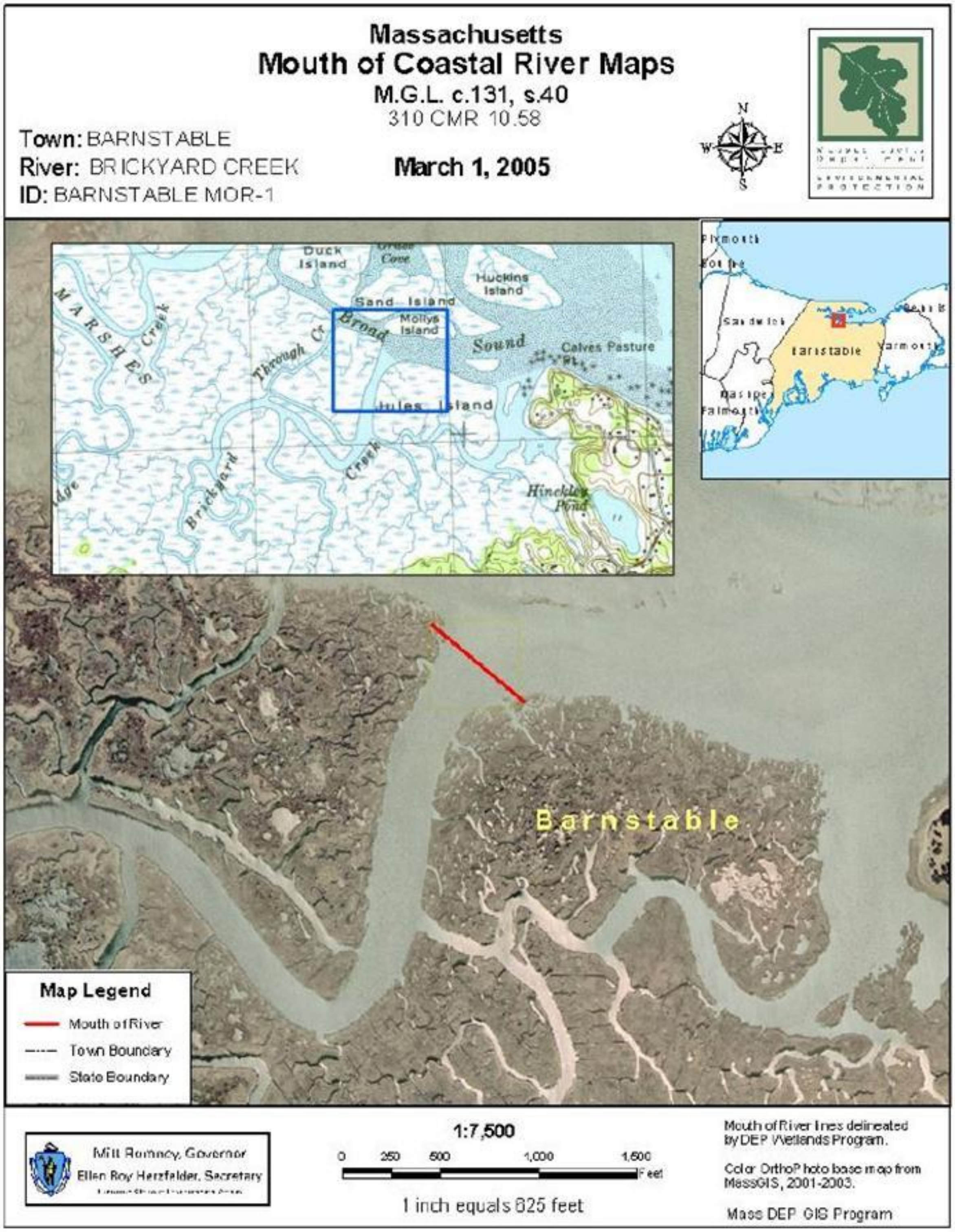


Figure II-6a. Mouth of Coastal Rivers designation for Brickyard Creek as determined by – MassDEP Wetlands Program. Estuaries on the down gradient reach of coastal rivers and major streams are part of the river system, and as such the mouth of the "river" is at the tidal inlet.

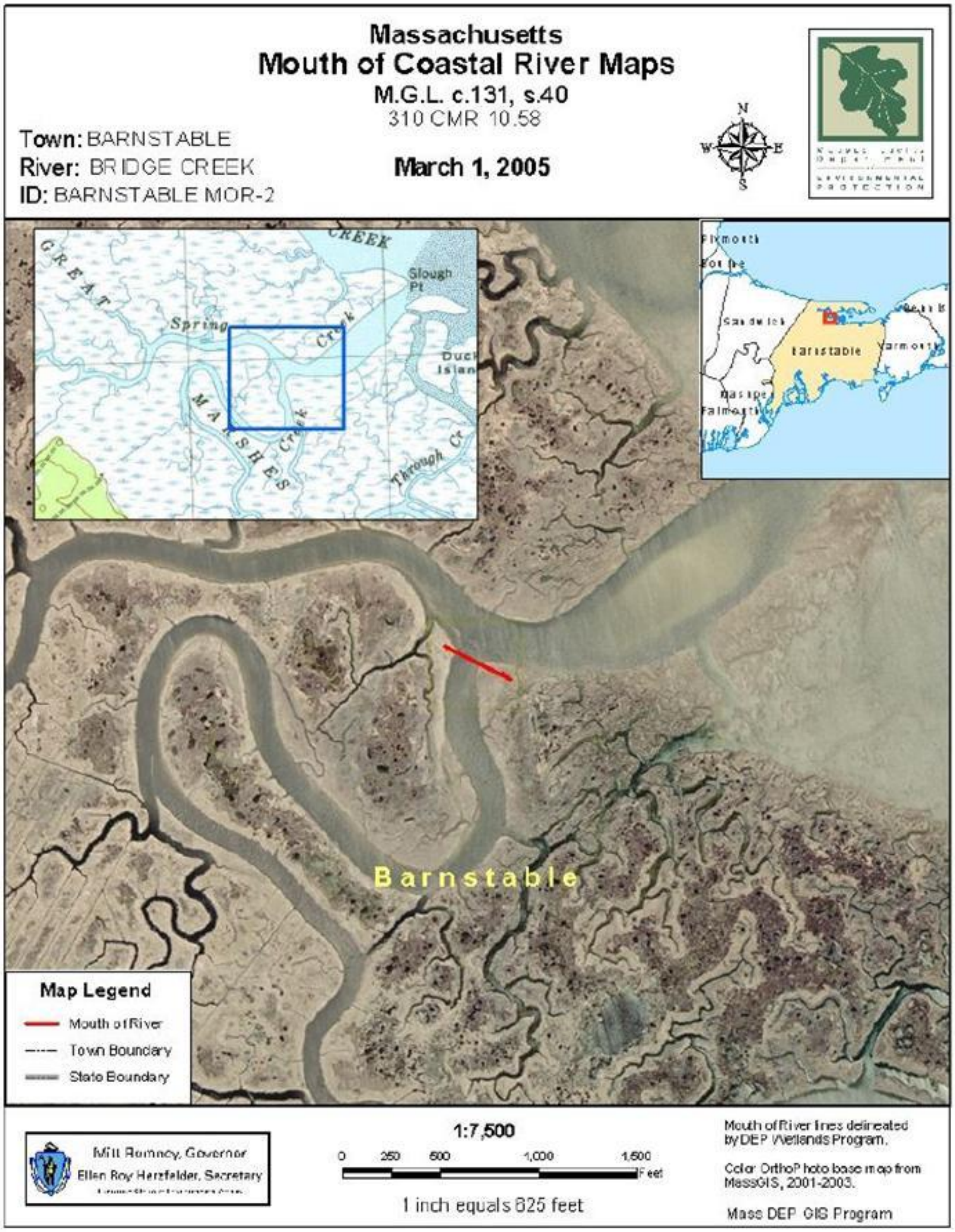


Figure II-6b. Mouth of Coastal Rivers designation for Bridge Creek as determined by – MassDEP Wetlands Program. Estuaries on the down gradient reach of coastal rivers and major streams are part of the river system, and as such the mouth of the "river" is at the tidal inlet.

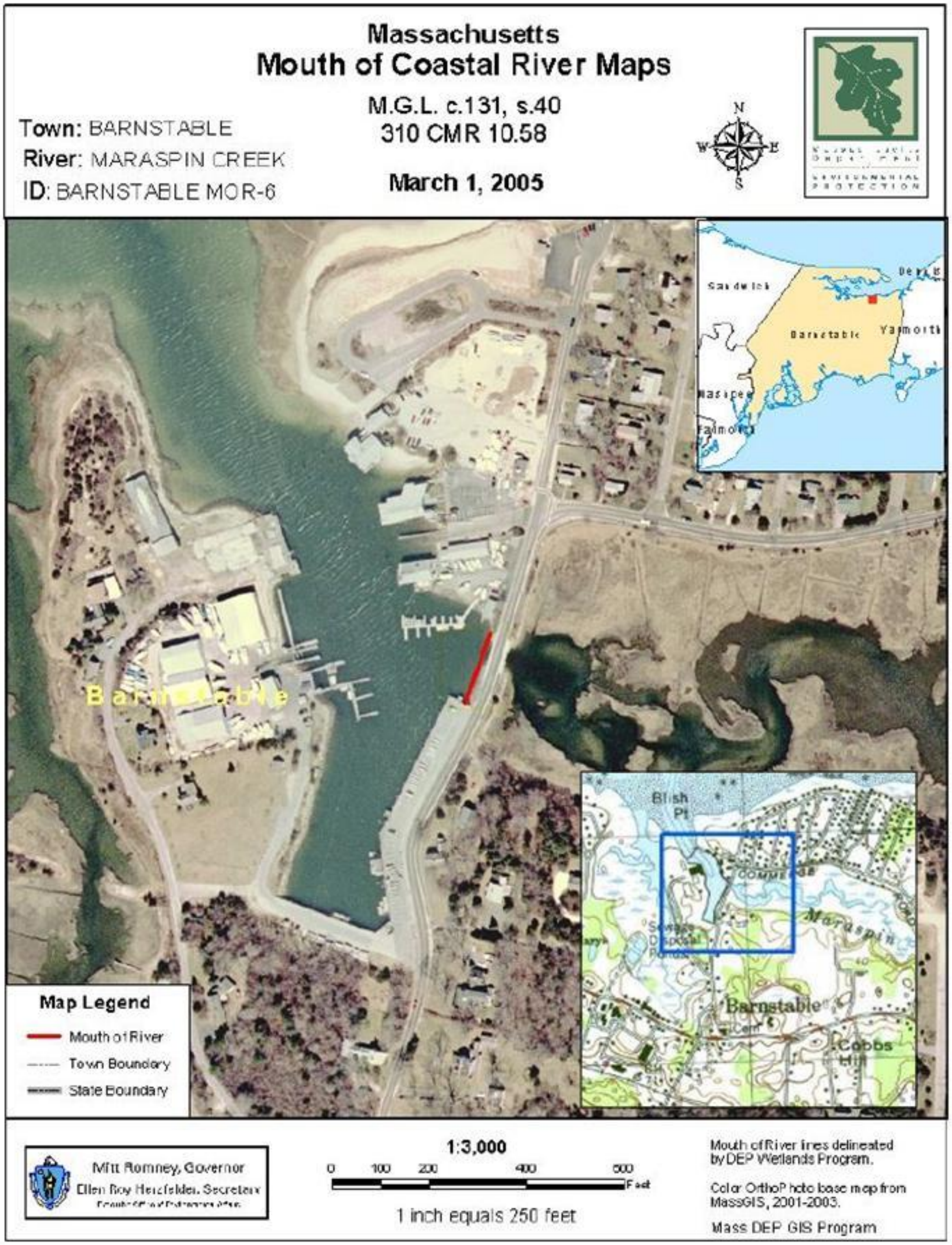


Figure II-6c. Mouth of Coastal Rivers designation for Maraspin Creek discharging to the Millway / Harbor from a south shore salt marsh as determined by – MassDEP Wetlands Program. Estuaries on the down gradient reach of coastal rivers and major streams are part of the river system, and as such the mouth of the "river" is at the tidal inlet.

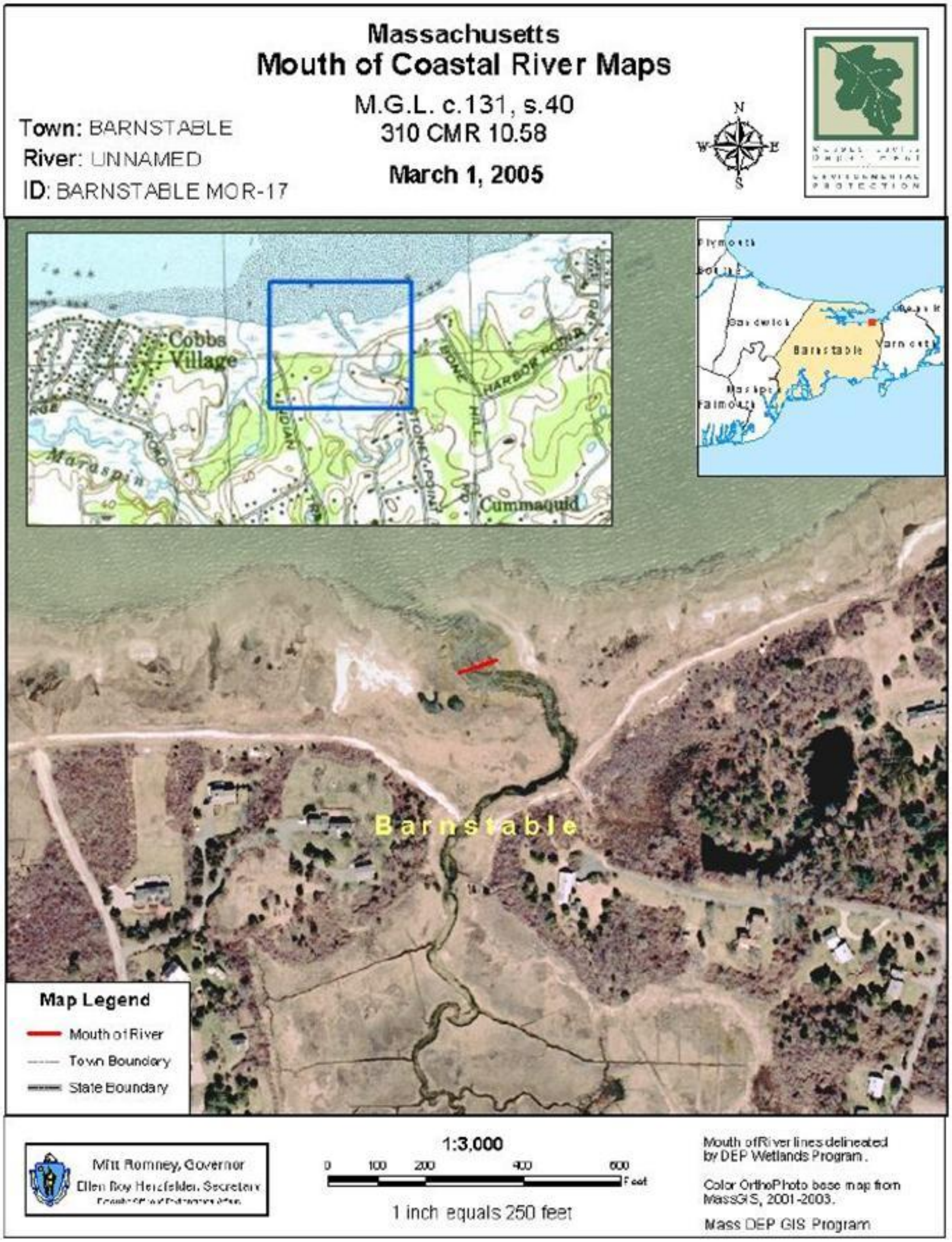


Figure II-6d. Mouth of Coastal Rivers designation for an un-named creek discharging to the harbor from a south shore salt marsh as determined by – MassDEP Wetlands Program. Estuaries on the down gradient reach of coastal rivers and major streams are part of the river system, and as such the mouth of the "river" is at the tidal inlet.

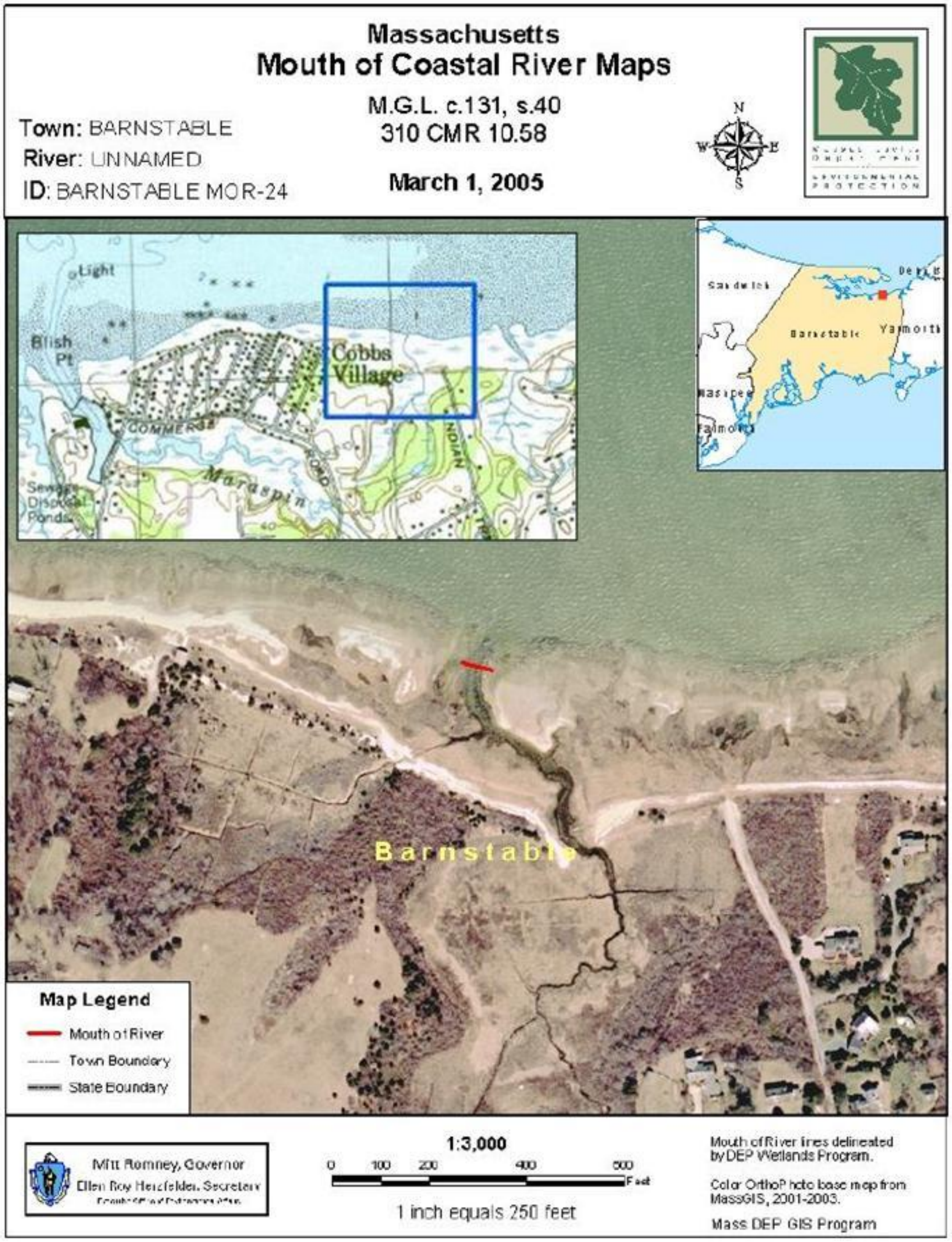


Figure II-6e. Mouth of Coastal Rivers designation for an un-named creek discharging to the harbor from a south shore salt marsh as determined by – MassDEP Wetlands Program. Estuaries on the down gradient reach of coastal rivers and major streams are part of the river system, and as such the mouth of the "river" is at the tidal inlet.

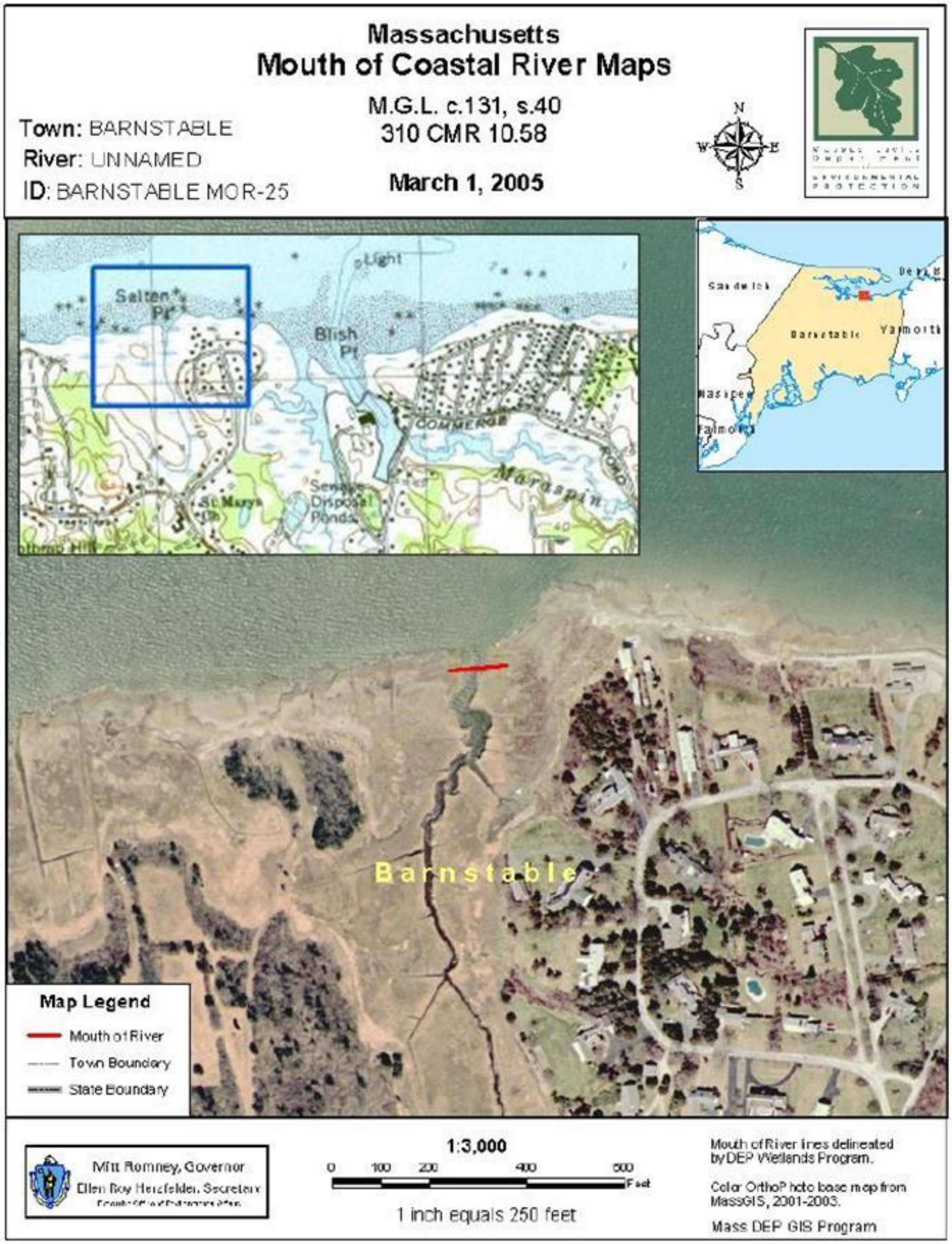


Figure II-6f. Mouth of Coastal Rivers designation for an un-named creek discharging to the harbor from a south shore salt marsh as determined by – MassDEP Wetlands Program. Estuaries on the down gradient reach of coastal rivers and major streams are part of the river system, and as such the mouth of the "river" is at the tidal inlet.

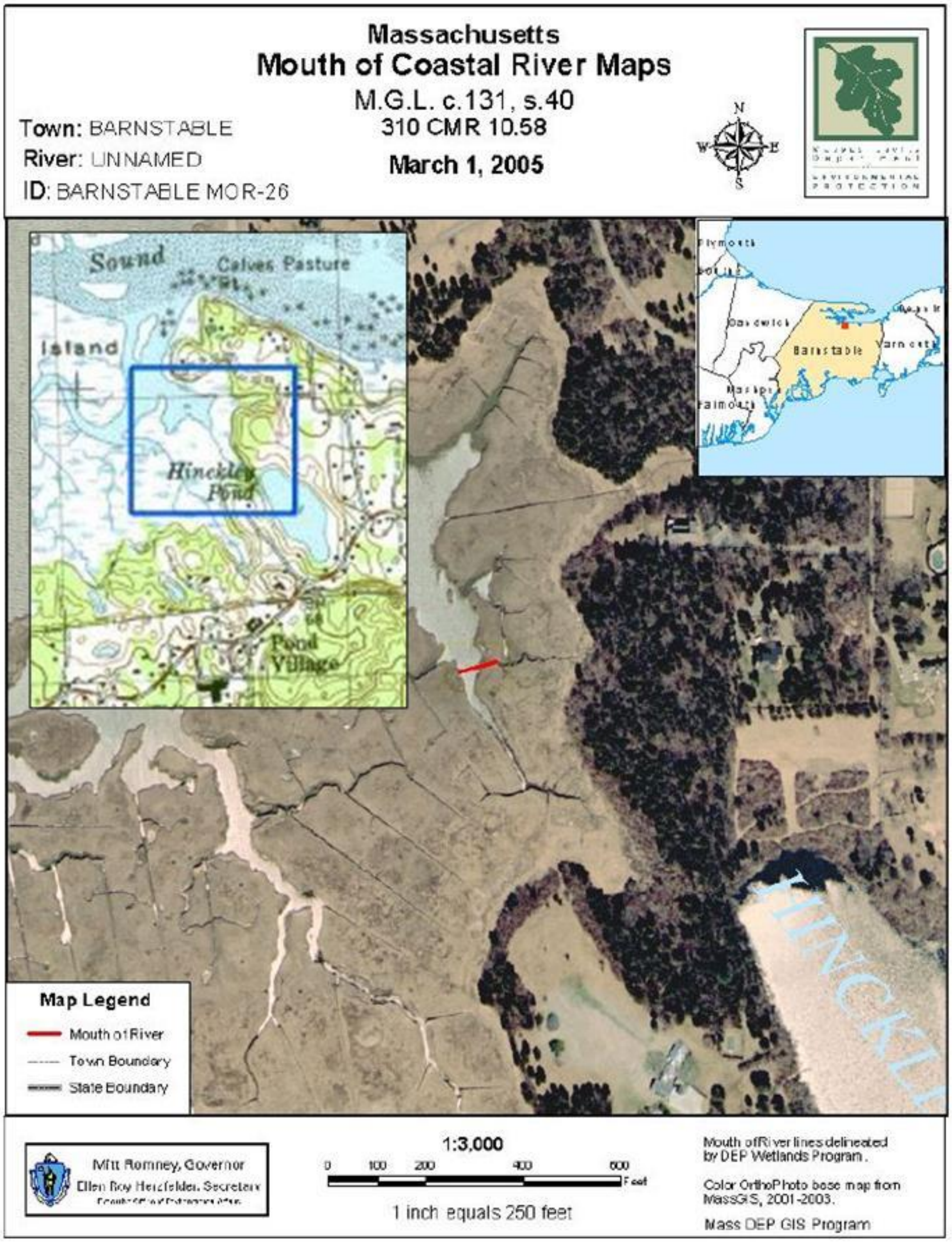


Figure II-6g. Mouth of Coastal Rivers designation for an un-named creek discharging from Hinckley Pond as determined by – MassDEP Wetlands Program. Estuaries on the down gradient reach of coastal rivers and major streams are part of the river system, and as such the mouth of the "river" is at the tidal inlet.

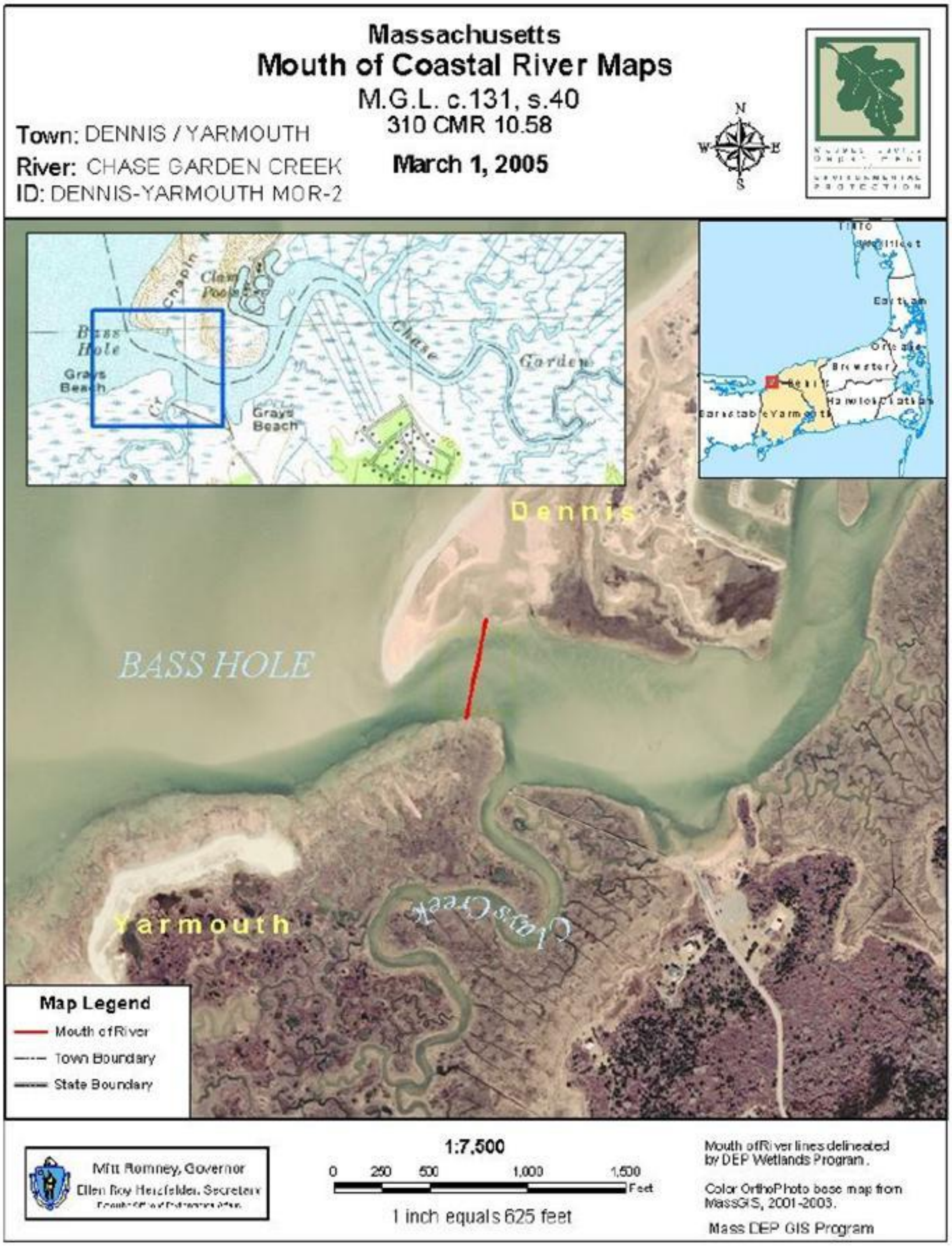


Figure II-6h. Mouth of Coastal Rivers designation for Bass Hole - Chase Garden Creek as determined by – MassDEP Wetlands Program. Estuaries on the down gradient reach of coastal rivers and major streams are part of the river system, and as such the mouth of the "river" is at the tidal inlet.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project team includes technical staff from the United States Geological Survey (USGS). The USGS groundwater modelers were central to the development of the groundwater modeling approach used by the Estuaries Project. The USGS has a long history of developing regional models for the six-groundwater flow cells on Cape Cod. Through the years, advances in computing, lithologic information from well installations, water level monitoring, stream flow measurements, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models. The MODFLOW and MODPATH models utilized by the USGS organize and analyze the available data using up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation. These questions include surface water/groundwater interactions, groundwater travel times, and drinking water well impacts that have also arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Barnstable Great Marshes. The Barnstable Great Marshes watershed is mostly split between the Towns of Barnstable, Yarmouth, and Dennis, but also includes a small portion of eastern Sandwich.

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to initially define the watershed or contributing area to the Barnstable Great Marshes under evaluation by the Project Team. The Barnstable Great Marshes estuarine system is a 16.5 square kilometer, shallow estuary with small channel and wide inlet linked to Cape Cod Bay. Numerous small streams flow into the main basin, including Alder Creek, Boat Cove Creek, Bridge Creek, and Chase Garden Creek. A number of smaller subestuaries also ring the main basin, including Bass Hole and the Millway/Barnstable Harbor. Watershed modeling was undertaken to sub-divide the overall watershed to the Barnstable Great Marshes system into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system including those listed, (b) defining contributing areas to major freshwater aquatic systems which attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining the land areas with groundwater travel times that are greater and less than 10 years time-of-travel to the estuary. These time-of-travel distributions within subwatersheds are used as a procedural check to gauge the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters at the time of the MEP analysis. The three-dimensional numerical groundwater model employed has also been used to evaluate the contributing areas to public water supply wells in the regional Sagamore and Monomoy flow cells of Cape Cod. The Barnstable Great Marshes watershed is located along the northeastern edge of the Sagamore groundwater lens and the northwestern edge of the Monomoy groundwater lens. USGS model outputs were also compared to surface water discharges measured as part of the MEP stream flow program (2005 to 2006), as well as some selective confirmation of 2009-2011 well pumping rates from the Yarmouth Water Department (personal communication, Dan Mills, YWD Superintendent).

The relatively transmissive sand and gravel deposits that comprise most of Cape Cod create a hydrologic environment where watershed boundaries are usually better defined by elevation of the groundwater and its direction of flow, rather than by land surface topography (Cambareri and Eichner 1998, Millham and Howes 1994a,b). Freshwater discharge to estuaries is usually composed of surface water inflow from streams, which receive much of their water from groundwater base flow, and direct groundwater discharge. For a given estuary, differentiating between these two water inputs and tracking the sources of nitrogen that they carry requires determination of the portion of the watershed that contributes directly to streams and the portion of the groundwater system that discharges directly into an estuary as groundwater seepage.

III.2 MODEL DESCRIPTION

Initial contributing areas to the Barnstable Great Marsh estuary system and its various subwatersheds, such as Bass Hole, Chase Garden Creek, and Bridge Creek, were delineated using the regional groundwater model of the Sagamore Lens and Monomoy Lens flow cells (Walter and Whealan, 2005). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al.*, 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the Barnstable Great Marshes system and its subwatersheds and also to determine portions of recharged water that may flow through fresh water ponds and streams prior to discharging into coastal water bodies.

The USGS Sagamore Flow Model grid consists of 246 rows, 365 columns and 20 layers, while the Monomoy Flow Model grid consists of 164 rows, 220 columns, and 20 layers. The horizontal model discretization, or grid spacing, in both portions of the model is 400 by 400 feet. The top 17 layers of the model in both flow cells extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below; the upper three layers of the Monomoy model are dry because the Monomoy has a lower elevation than the overlapping Sagamore groundwater model that was constructed at the same time (Walter and Whealan, 2005). Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 519 feet below NGVD 29 in the Sagamore Lens and up to 525 feet below NGVD 29 in the Monomoy). In the Barnstable Great Marshes watershed area, the groundwater model included bedrock at depths approximately 200 to 300 feet below NGVD 29 (Walter and Whealan, 2005). In the groundwater flow model, these bedrock depths mean that the lowest model layer is inactive throughout most of the watershed area. The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, was used to simulate the top of the water table, which also varies in elevation depending on the location within the lens.

Direct rainwater run-off in these Cape Cod aquifer materials is typically rather low. Lithological data used to determine hydraulic conductivities used in the groundwater model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters in the groundwater models were determined through calibration to observed water levels and available stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and stream flow data collected in 1989-1990 as well as 2003.

The glacial sediments that comprise the aquifer of the both the Sagamore Lens and the Monomoy Lens consist of gravel, sand, silt, and clay that were deposited 15,000 to 16,000 years ago in a variety of depositional environments by continental ice sheets. The Barnstable Great Marshes watershed area generally consists of Lake Deposits bracketed along the southern edge by the Sandwich Moraine (Walter and Whealan, 2005). The Sandwich Moraine was formed when the regional Cape Cod Bay ice lobe re-advanced to the south and excavated and piled up previously deposited materials. The edge of the ice sheet eventually retreated north to a location within the current Cape Cod Bay and a large lake formed between the ice sheet and the moraine. This Lake Cape Cod Bay trapped fine sediments and clays flowing off the face of the ice sheet along its bottom. These sediments generally underlie the extensive saltwater marshes located along the current northern shoreline of Cape Cod.

Although these are glacial materials vary, modeling and field measurements of contaminant transport at the Massachusetts Military Reservation have shown that groundwater flowpaths are largely unaffected by the transitions between outwash and moraine materials (*e.g*., Masterson, *et al*., 1997). Most of the lake bottom deposit areas along the northern portion are covered by saltwater marshes, but the presence of extensive streams at margin of the marshes suggests that large portions of the upgradient aquifer is discharging along this margin. This is largely supported by the good agreement between stream watershed flows estimated in the groundwater model and MEP streamflows measured at the gauged streams (see Section IV.2). This agreement is also consistent with similar comparisons in other marsh-dominated systems along the northern coast of Cape Cod (*e.g.*, Howes, *et al*., 2007, Howes, *et al.*, 2015).

The regional USGS Sagamore Lens groundwater model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of 27.25 inches/year and the pumping of public-supply wells at average annual withdrawal rates for the period 1995-2000 with a 15% consumptive loss. This recharge rate is based on the most recent USGS information. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. After accounting for the consumptive loss, water withdrawn from the modeled aquifer by public drinking water supply wells is evenly returned within residential areas designated as using on-site septic systems.

III.3 BARNSTABLE GREAT MARSHES SYSTEM CONTRIBUTORY AREAS

The initial refined watershed and sub-watershed boundaries for the Barnstable Great Marshes embayment system, including the ponds, streams, and water supply wells were modeled by the United States Geological Survey (USGS). Model outputs of the watershed boundaries are usually presented as “saw toothed” lines that reflect the movement of modeled particles between the grid cells that make up the groundwater model and how those cells are organized to reflect natural features, such as pond shorelines, wetlands, river segments, and contributing areas to public water supplies. In order utilize the guidance provided by the model, these modeled lines were “smoothed” to: (a) correct for the model grid spacing, (b) to enhance the accuracy of the characterization pond and coastal shorelines, (c) to include water table data in the lower regions of the watersheds near the coast (as available), (d) to more closely match the sub-estuary segmentation of the tidal hydrodynamic model and (e) to address streamflow measurements collected as part of the MEP. The smoothing refinement process was a collaborative effort developed between the USGS and the rest of the MEP Technical Team. The MEP sub-watershed delineation includes 10-yr time-of-travel boundaries, subwatersheds to public water supply wells, and MEP stream gauges within the overall watershed. The smoothing simplification of watershed delineations/recharge areas lines and other model outputs usually involved visual curve fitting and checking model outputs against aerial maps of ponds, streams, and wetlands. MEP staff also added “subwatersheds” within the marsh shield in order to more accurately characterize the spatial distribution of freshwater inputs to the marsh. These delineations were based on natural segmentation within the marsh and used the natural channels as guides. These are not watersheds in the typical sense because water levels during flood tides cover the whole marsh, but these are a reasonable basis for refining the distribution of freshwater inputs to the marsh.

Overall, 45 sub-watershed areas were delineated within the Barnstable Great Marshes study area (Figure III-1).

Table III-1 provides the daily freshwater discharge volumes for the subwatersheds as calculated from the MEP watersheds; these volumes were used in the salinity calibration of the MEP water quality model and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to the directly measured surface water discharges. The overall estimated freshwater flow into the Barnstable Great Marshes system from the MEP delineated watershed is 171,829 m3/d. This flow includes balanced corrections for flow added to the watershed by the shared, adjacent ponds, such as Lawrence Pond in Sandwich (Howes, *et al*., 2006) and Shallow Pond in Barnstable (Eichner, 2009). Comparison of modeled watershed flow to measured flow at the six MEP gauges, which collected streamflow measurements between September 2006 and August 2007, were within 10% of each other (see Section IV.2). The measured flows are used for calibration of the estuarine water quality model.

The MEP watershed delineation is the second watershed delineation completed in recent years for Barnstable Great Marshes System. Figure III-2 compares the delineation completed under the current effort with the delineation previously completed by the Cape Cod Commission (Eichner, *et al*., 1998). The CCC delineation was largely based on local and regional water table measurements collected from available groundwater elevation data collected over a number of years, including some of the same data used in the USGS groundwater model, and normalized to average conditions. The Commission's delineation was incorporated into the Commission's regulations through the four versions of the Regional Policy Plan (CCC, 1996, 2001, 2009, and 2012).

The MEP watershed area for the Barnstable Great Marshes System as a whole is 6% larger than 1998 CCC delineation (21,347 acres vs. 20,174 acres, respectively). Comparison of estuary surface shows that the MEP area is 32% larger, mostly due to inclusion of more water within the main inlet. The small watershed area differences are mostly in the land areas included near the main inlet and a slight shift of the regional groundwater divide between Cape Cod Bay and Vineyard Sound further to the south near Dennis and Greenough Pond in Yarmouth. The MEP watershed delineation also includes much more refined interior sub-watersheds to various components of the Barnstable Great Marshes estuarine system, such as selected ponds and streams that were not included in the CCC delineation. The inner subwatershed delineations show the connections between adjacent watersheds and the complexities of flow paths. These refinements are another benefit of the update of the regional groundwater model (Walter and Whealan, 2005).

The MEP watersheds compared favorably with both measured streamflow and pumping rates from public water supply wells. MEP staff gathered streamflow in six streams that flow into various portions of the main system (detailed in Section IV.2). Modeling results were also checked against recent pumping rates from public water supplies within the watershed, as well as where this water was distributed within the overall watershed. The majority of water pumped from Barnstable Fire District wells (watershed #15) is returned to development within the watershed based on the District's service area (personal communication, Thomas Rooney, Superintendent, September 2015). However, a portion of the pumped water (~27% based on average 2006-2008 water use) is removed from the watershed by the limited sewer collection area near Barnstable Village. Similarly, MEP staff reviewed 2009-2011 pumping rates from the Yarmouth wellfield (watershed #34) to address measured MEP flow in Whites Brook (personal communication, Dan Mills, Superintendent, October 2015). This review found that the actual, lower reported pumping rates provided a reasonable balance between the measured and modeled MEP watershed/stream flows. These types of comparison between measured and modeled information is another step to reinforce the reliability of the watershed delineations.

The evolution of the watershed delineations for the Barnstable Great Marshes system has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected; the model allows all this data to be organized and to be brought into congruence with adjacent watersheds. The evaluation of older data and incorporation of new data during the development of the model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model and the use of this model for the development and evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the down gradient estuary. The MEP watershed delineation was used to develop the watershed nitrogen loads to each of the aquatic systems and ultimately to the estuarine waters of the Barnstable Great Marshes system (Section V.1).

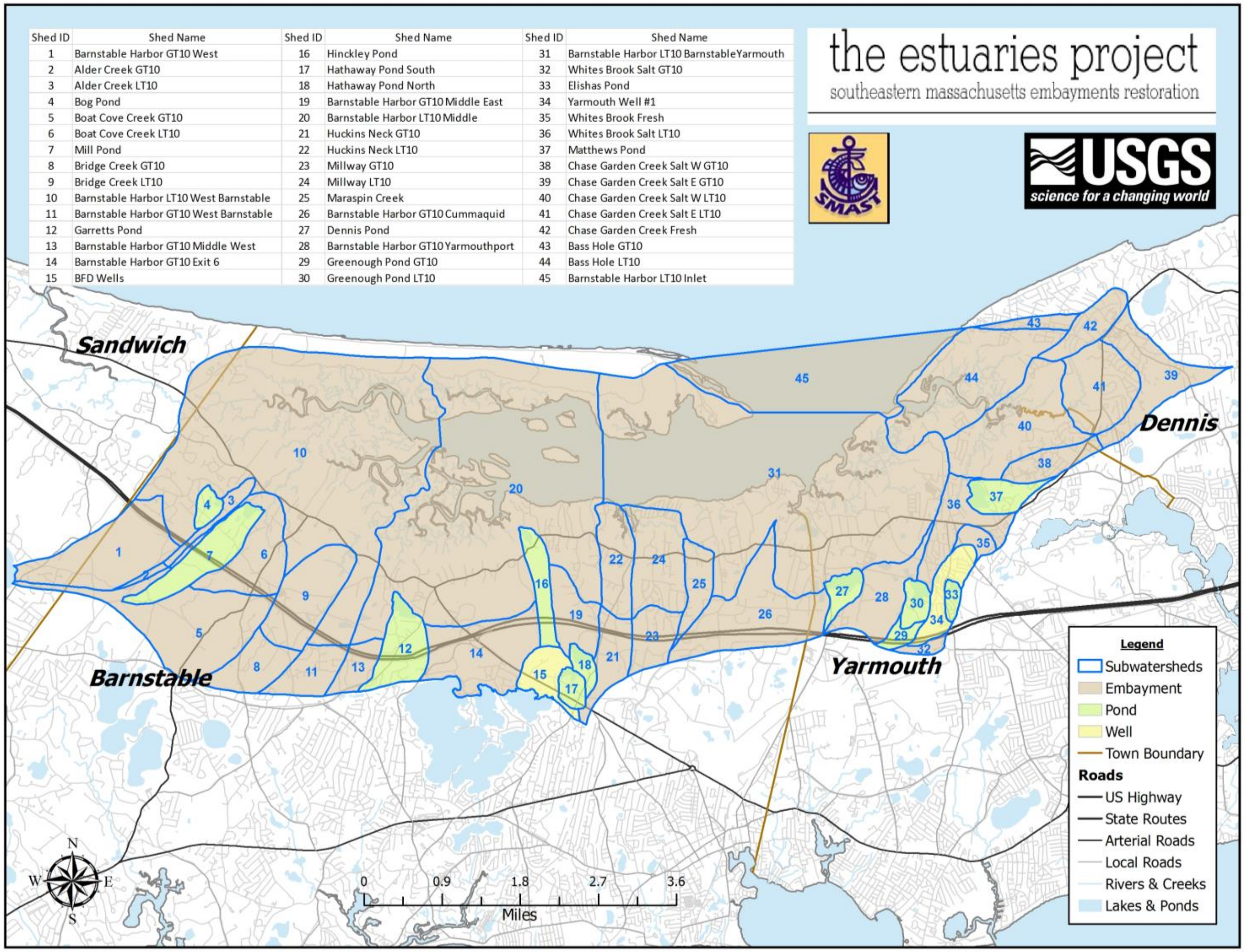


Figure III-1. Watershed delineation for the Barnstable Great Marshes estuary system. This estuary system exchanges tidal waters with Cape Cod Bay. Subwatershed delineations are based on USGS groundwater model output with modifications to better address pond and estuary shorelines and MEP stream gauge measurements. Ten-year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names.

Table III-1. Daily groundwater discharge from each of the sub-watersheds in the watershed to the Barnstable Great Marshes system estuary, as determined from the MEP watersheds and corrected return flow from public water supply wells.

Watershed	#	Watershed Area (acres)	% contributing to Estuaries	Discharge	
				m^3/day	ft^3/day
BarnHbr GT10W	1	659	100%	5,056	178,538
Alder Creek GT10	2	22	100%	172	6,074
Alder Creek LT10	3	86	100%	657	23,219
Bog Pond	4	67	100%	514	18,165
Boat Cove Creek GT10	5	848	100%	6,509	229,870
Boat Cove Creek LT10	6	435	100%	3,339	117,910
Mill Pond	7	337	100%	2,587	91,366
Bridge Creek GT10	8	269	100%	2,068	73,028
Bridge Creek LT10	9	469	100%	3,600	127,118
BarnHbr LT10 West Barnstable	10	3,753	100%	28,800	1,017,063
BarnHbr GT10 West Barnstable	11	229	100%	1,760	62,160
Garretts Pond	12	298	100%	2,285	80,685
BarnHbr GT10 MidW	13	153	100%	1,176	41,519
BarnHbr GT10 - Exit 6	14	588	100%	4,514	159,425
BFD Wells	15	176	100%	1,353	47,777
Hinckley Pond	16	180	100%	1,382	48,819
Hathaway Pond S	17	78	100%	600	21,202
Hathaway Pond N	18	65	100%	499	17,628
Barnstable Hbr GT10 MidE	19	255	100%	1,956	69,069
Barnstable Hbr LT10 Mid	20	2,966	100%	22,763	803,852
Huckins Neck GT10	21	230	100%	1,765	62,336
Huckins Neck LT10	22	272	100%	2,087	73,686
Millway GT10	23	216	100%	1,656	58,473
Millway LT10	24	424	100%	3,255	114,944
Maraspin Creek	25	182	100%	1,397	49,327
BarnHbr GT10 Cummaquid	26	736	100%	5,652	199,587
Dennis Pond	27	103	100%	792	27,956
BarnHbr GT10 - Yarmouthport	28	356	100%	2,734	96,550
Greenough Pond GT10	29	59	100%	453	16,011
Greenough Pond LT10	30	95	100%	726	25,646
BarnHbr LT10 - BarnYarm	31	2,736	100%	20,996	741,465
Whites Brook Salt GT10	32	45	100%	346	12,219
Elishas Pond	33	46	100%	349	12,341

Table III-1 (continued). Daily groundwater discharge from each of the sub-watersheds in the watershed to the Barnstable Great Marshes system estuary, as determined from the MEP watersheds and corrected return flow from public water supply wells.

Yarm_Well1	34	194	100%	1,488	52,534
Whites Brook Fresh	35	71	100%	548	19,348
Whites Brook Salt LT10	36	316	100%	2,427	85,693
Matthews Pond	37	167	100%	1,284	45,361
Chase Garden Crk Salt W GT10	38	188	100%	1,443	50,966
Chase Garden Creek Salt E GT10	39	439	100%	3,369	118,981
Chase Garden Crk Salt W LT10	40	729	100%	5,596	197,606
Chase Garden Crk Salt E LT10	41	454	100%	3,484	123,046
Chase Garden Creek Fresh	42	205	100%	1,574	55,588
Bass Hole GT10	43	86	100%	664	23,435
Bass Hole LT10	44	955	100%	7,327	258,747
Barnstable Hbr LT10 Inlet	45	106	100%	817	28,835
Additions from outside watershed			100%	8,012	282,934
TOTAL BARNSTABLE GREAT MARSHES SYSTEM				171,829	6,068,104

Notes:
1) Discharge volumes are based on 27.25 inches of annual recharge on watershed areas.
2) Watershed flow is added to the system from the watersheds to Lawrence Pond, Lake Wequaquet, and Shallow Pond. These ponds are located along the watershed boundary and add flow to a number of watersheds.
3) listed flows do not include precipitation on the surface of the estuary
4) totals may not match due to rounding.

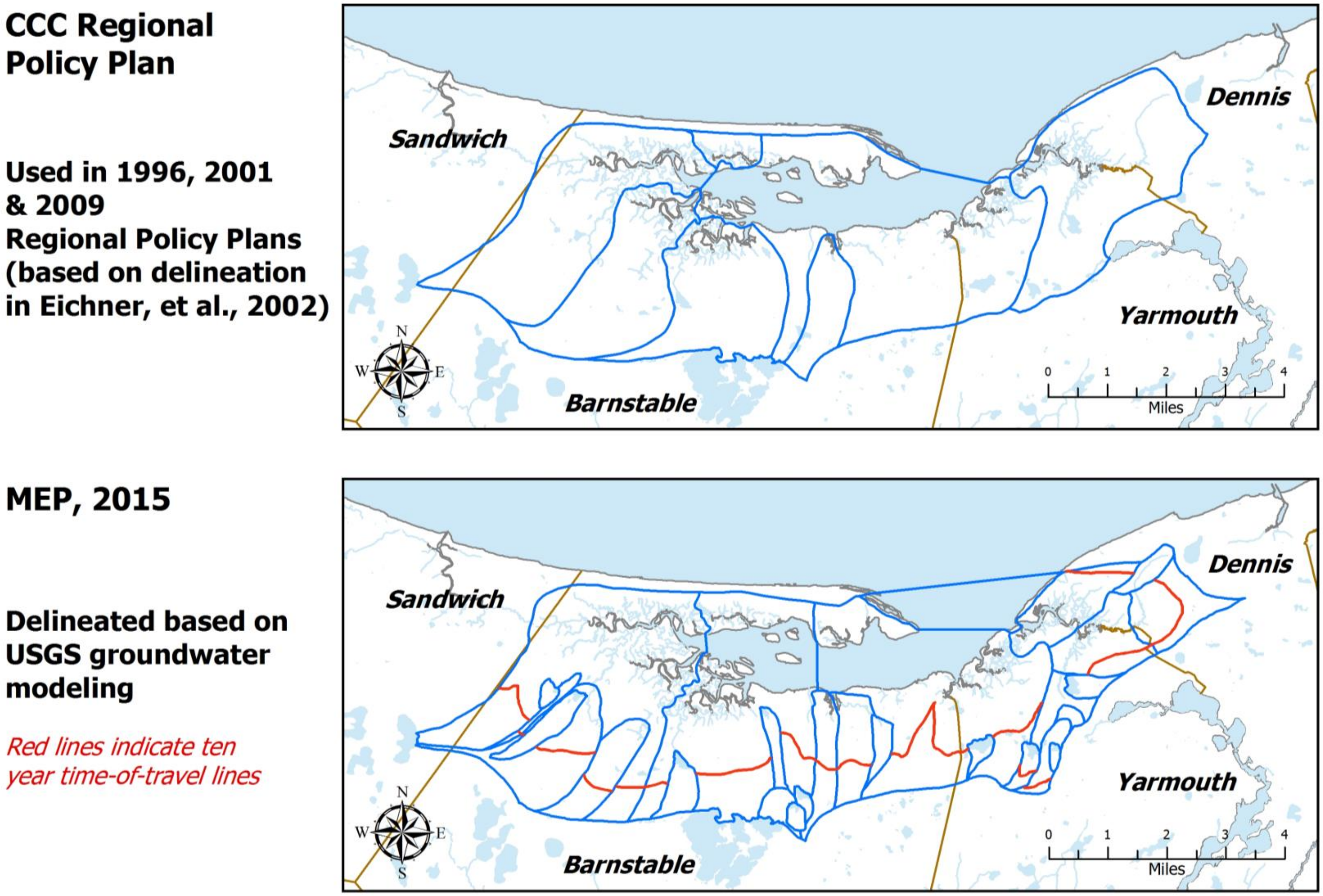


Figure III-2. Comparison of MEP Barnstable Great Marshes watershed and sub-watershed delineations used in the current assessment and the Cape Cod Commission watershed delineation (Eichner and others, 1998), which had been used in three Barnstable County Regional Policy Plans (CCC, 1996, 2001, 2009). The MEP watershed area for the Barnstable Great Marshes system as a whole is 6% larger than CCC delineation. Barnstable Great Marshes exchanges tidal waters with Cape Cod Bay to the north.

IV. WATERSHED NITROGEN LOADING TO EMBAYMENT: LAND USE, STREAM INPUTS, AND SEDIMENT NITROGEN RECYCLING

IV.1 WATERSHED LAND USE BASED NITROGEN LOADING ANALYSIS

Management of nutrient related water quality and habitat health in coastal waters requires determination of the amount of nitrogen transported by freshwaters (surface water flow, groundwater flow) from the surrounding watershed to the receiving embayment of interest. In southeastern Massachusetts, the nutrient of management concern for estuarine systems is nitrogen and this is true for the Barnstable Great Marsh estuary system. Determination of watershed nitrogen inputs to these embayment systems requires the (a) identification and quantification of the nutrient sources and their loading rates to the land and aquifer, (b) confirmation that a groundwater transported load has reached the embayment at the time of analysis, and (c) quantification of nitrogen attenuation that can occur during travel through lakes, ponds, streams and marshes prior to reaching the estuary. This latter natural attenuation process results from biological processes that naturally occur within these ecosystems. Failure to account for attenuation of nitrogen during transport results in an over-estimate of nitrogen inputs to an estuary and an underestimate of the sensitivity of a system to new inputs (or removals). In addition to the nitrogen transport from land to sea, the amount of direct atmospheric deposition on each embayment surface must be determined as well as the amount of nitrogen recycling within the embayment, specifically nitrogen regeneration from sediments. Sediment nitrogen recycling results primarily from the settling and decay of phytoplankton and macroalgae (and eelgrass when present). During decay, organic nitrogen is transformed to inorganic forms, which may be released to the overlying waters or lost to denitrification within the sediments. Permanent burial of nitrogen in the sediments is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters or in some cases a sink for nitrogen reaching the bottom. Failure to include the nitrogen balance of estuarine sediments and the watershed attenuation generally leads to errors in predicting water quality, particularly in determination of summertime nitrogen load to embayment waters.

In order to determine watershed nitrogen loading inputs to the Barnstable Great Marsh estuary system, the MEP Technical Team developed nitrogen-loading rates (Section IV.1) to each component of the estuary and its watersheds (Section III). The Barnstable Great Marsh watershed was sub-divided to define contributing areas or subwatersheds to each of the major inland freshwater systems and to each major component basin of the estuary. Further sub-divisions were made to identify watershed areas where a nitrogen discharge reaches estuary waters in less than 10 years or greater than 10 years. A total of 45 subwatersheds were delineated in the overall Barnstable Great Marsh watershed, including watersheds to six MEP gauged streams, 10 ponds, and 2 public water supply wellfields (see Chapter III). The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each portion of the estuary.

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the estuary. This review involves a temporal review of land use changes, the time of groundwater travel provided by the USGS watershed model, and review of data at natural collection points, such as streams and ponds. Evaluation and delineation of ten-year time of travel zones are a regular part of the MEP watershed analysis. Ten-year time of travel subwatersheds in the Barnstable Great Marsh watershed have been delineated for ponds, streams and the estuary itself. Review of less than 10 year (LT10) and greater than 10 year travel time (GT10) watersheds indicates that 66% of the unattenuated nitrogen load from the whole watershed is within less than 10 year travel time to the estuary (Table IV-1). When nitrogen loads from precipitation direct to the estuary surface are added to the LT10 total, the percentage of the overall system load within less than 10 years increases to 73% without adjusting for age of dwellings or public water supply redistribution. The overall review includes refinements for transfer of loads from shared pond subwatersheds: a) Lake Wequaquet, Bearses Pond, and Shallow Pond shared with the Centerville River estuary (Howes, *et al.*, 2006a) and b) Lawrence Pond shared with the Scorton Creek estuary (Howes, *et al*., 2015) and the Three Bays estuary (Howes, *et al.*, 2006b).

This analysis can become complicated by movement of water and nitrogen loads by public drinking water supply and the age of houses/buildings within the GT10 subwatersheds. For example, the Barnstable Fire District (BFD) service area is primarily within subwatersheds less than 10 years of travel time from the estuary, but the wellfield and its contributing area is located within the greater than 10 year delineation. It would be reasonable to add all the nitrogen load from the BFD wellfield watershed (subwatershed #15), including contributions for Hathaway Pond South and Shallow Pond to the LT10 nitrogen load totals. In addition, MEP staff also reviewed the age of single family residences in the greater than 10 year subwatersheds to assess whether nitrogen loads from these subwatersheds are likely to have already reached the estuary. This review of year-built information in the town assessor's database generally indicated that average age of single family residences (the predominant land use) were greater than 20 years. For example, within subwatershed #14 (Barnstable Harbor GT10 Exit 6) the 163 single family residential parcels (unsplit) were constructed an average of 28 years before the MEP stream sampling period (2006-2007).

The overall conclusion from the analysis of the timing of development relative to groundwater travel times (including the GT10 subwatersheds) is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuary (after accounting for natural attenuation, see below). Additionally, the distinction between time of travel in the subwatersheds is not important for modeling existing watershed nitrogen loading conditions. Based on the review of all of this information, it was determined that the Barnstable Great Marsh estuary is currently in balance with its watershed load.

In order to determine nitrogen loads from the watersheds, detailed individual lot-by-lot data is used for some portion of the loads, while information developed from other detailed site-specific studies is applied to other portions. The Linked Watershed-Embayment Management Model (Howes, *et al.*, 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land uses and pre-determined nitrogen loading rates based on regional analyses. For the Barnstable Great Marsh Estuarine System, the model used Town of Barnstable, Town of Yarmouth, Town of Dennis, and Town of Sandwich land-use data that is transformed into nitrogen loads using both regional nitrogen loading factors and local watershed-specific data (such as parcel-by-parcel municipal water use). Determination of the nitrogen loads required obtaining watershed-specific information regarding wastewater, fertilizers, runoff from impervious surfaces and atmospheric deposition. The primary regional factors were derived for southeastern Massachusetts from direct measurements. The resulting nitrogen loads represent the "potential" or unattenuated nitrogen load to each receiving estuarine basin, since attenuation within surface waters (streams, ponds, wetlands) is included at a later stage.

Table IV-1. Percentage of unattenuated nitrogen loads in less than ten year time-of-travel subwatersheds to Barnstable Great Marsh.

WATERSHED		LT10	GT10	TOTAL	%LT10
Name	#	kg/yr	kg/yr	kg/yr	
BarnHbr GT10W	1		1,419	1,419	0%
Alder Creek GT10	2		7	7	0%
Alder Creek LT10	3	384		384	100%
Bog Pond	4	473		473	100%
Boat Cove Creek GT10	5		1,283	1,283	0%
Boat Cove Creek LT10	6	1,171		1,171	100%
Mill Pond	7	1,005		1,005	100%
Bridge Creek GT10	8		1,303	1,303	0%
Bridge Creek LT10	9	1,224		1,224	100%
BarnHbr LT10 West Barnstable	10	6,171		6,171	100%
BarnHbr GT10 West Barnstable	11		1,392	1,392	0%
Garretts Pond	12	1,587		1,587	100%
BarnHbr GT10 MidW	13		668	668	0%
BarnHbr GT10 - Exit 6	14		2,878	2,878	0%
BFD Wells	15		1,023	1,023	0%
Hinckley Pond	16	373		373	100%
Hathaway Pond S	17		338	338	0%
Hathaway Pond N	18		117	117	0%
Barnstable Hbr GT10 MidE	19		327	327	0%
Barnstable Hbr LT10 Mid	20	4,605		4,605	100%
Huckins Neck GT10	21		590	590	0%
Huckins Neck LT10	22	559		559	100%
Millway GT10	23		758	758	0%
Millway LT10	24	2,027		2,027	100%
Maraspin Creek	25	1,360		1,360	100%
BarnHbr GT10 Cummaquid	26		2,538	2,538	0%
Dennis Pond	27		455	455	0%
BarnHbr GT10 - Yarmouthport	28		554	554	0%
Greenough Pond GT10	29		28	28	0%
Greenough Pond LT10	30		164	164	0%
BarnHbr LT10 - BarnYarm	31	9,500		9,500	100%
Whites Brook Salt GT10	32		141	141	0%
Elishas Pond	33	489		489	100%
Yarm_Well1	34	1,769		1,769	100%
Whites Brook Fresh	35	715		715	100%
Whites Brook Salt LT10	36	3,256		3,256	100%
Matthews Pond	37	1,024		1,024	100%
WATERSHED		LT10	GT10	TOTAL	%LT10
Name	#	kg/yr	kg/yr	kg/yr	

Table IV-1. Percentage of unattenuated nitrogen loads in less than ten year time-of-travel subwatersheds to Barnstable Great Marsh.

Chase Garden Crk Salt W GT10	38		1,807	1,807	0%
Chase Garden Creek Salt E GT10	39		4,303	4,303	0%
Chase Garden Crk Salt W LT10	40	3,289		3,289	100%
Chase Garden Crk Salt E LT10	41	4,209		4,209	100%
Chase Garden Creek Fresh	42	1,561		1,561	100%
Bass Hole GT10	43		914	914	0%
Bass Hole LT10	44	1,790		1,790	100%
Barnstable Hbr LT10 Inlet	45	30		30	100%
ponds shared with other watersheds			1,883	1,883	0%
Barnstable Great Marsh Whole System Watershed		48,571	24,890	73,461	66%
Estuary Surface		18,265		18,265	100%
Barnstable Great Marsh Whole System		66,835	24,890	91,726	73%

Notes:
a) Whole system totals may not add due to rounding.
b) Review of public drinking water supply service areas suggest that most of the Barnstable Fire District wells subwatershed nitrogen load (shed #15) should be considered as reaching the estuary in less than 10 years.
c) Review of year-built information for single-family residences within the GT10 subwatersheds (the predominant parcel type in the overall watershed) shows that the average age of residences within the GT10 subwatersheds is generally older than 20 years. This information suggests that average nitrogen loads from the GT10 subwatersheds are also currently reaching the estuary and when combined with the groundwater time-of-travel info confirms that estuarine water quality data is in balance with overall watershed nitrogen loading.

Natural attenuation of nitrogen during transport from land-to-sea within the Barnstable Great Marsh watershed was determined based upon site-specific studies of each major stream discharging to this estuary and assumed attenuation in the upgradient freshwater ponds. Streamflow was characterized at gauge locations from: Alder Creek, Boat Cove Creek, Bridge Creek, Maraspin Creek, White's Brook, and the freshwater portion of Chase Garden Creek. Land-use analysis of the contributing areas to these stream discharge points allowed comparisons between field collected nitrogen load data from the streams and estimates from the nitrogen-loading sub-model. Nitrogen attenuation in individual ponds is generally estimated based on available information. Attenuation through the ponds is conservatively assumed to equal 50% unless available water column monitoring and pond physical data is reliable enough to calculate a pond-specific attenuation factor. Streamflow and associated surface water attenuation is included in the MEP's nitrogen attenuation and freshwater flow investigation, presented in Section IV.2.

Natural attenuation during stream transport or in passage through fresh ponds of sufficient size to effect groundwater flow patterns (area and depth) is a standard part of the data collection effort of the MEP. In the present effort, ten freshwater ponds have delineated subwatersheds within the Barnstable Great Marsh watershed: Bog Pond, Mill Pond, Garretts Pond, Dennis Pond, Hathaway Pond S, Hathaway Pond N, Elishas Pond, Matthews Pond, Hinckley Pond, and Greenough Pond. In addition, delineation was undertaken for four other ponds that are mostly outside of the watershed but are shared with the Barnstable Great Marsh watershed: Lawrence Pond, Lake Wequaquet, Bearses Pond, and Shallow Pond. If smaller aquatic features that have not been included in this MEP analysis were providing additional attenuation of nitrogen, nitrogen loading to the estuary would only be slightly (~10%) overestimated given the distribution of nitrogen sources within the watershed.

Based upon the evaluation of the watershed systems, the MEP Technical Team used the Nitrogen Loading Sub-Model to estimate nitrogen loading for the subwatersheds that directly discharge groundwater to the estuary without flowing through one of these interim pond and stream measuring points. The direct discharge subwatershed were combined with subwatersheds with surface freshwater attenuation determined from field measurements to select appropriate nitrogen attenuation rates to calculate total watershed loading to the estuary. Internal nitrogen recycling was also determined throughout the tidal reaches of the Barnstable Great Marsh Estuarine System; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Water Use Database Preparation

Since the watershed to the Barnstable Great Marsh Estuary is shared among the towns of Barnstable, Yarmouth, Dennis, and Sandwich, Estuaries Project staff obtained digital parcel and tax assessor's data from the town to serve as a base for the watershed nitrogen loading model. Digital parcels and land use/assessors data are from 2011, 2012, 2012, and 2011, respectively. Using GIS techniques, this data was linked to at least three years of individual account water use data from the various water suppliers: Barnstable Fire District, 2006-2008; Hyannis, 2006-2011, Dennis, 2007-2009; Yarmouth, 2005-2011, and Sandwich, 2007-2010. The resulting unified watershed database contains traditional information regarding land use classifications (MassDOR, 2015) plus additional information developed by the towns. The database matching efforts were completed with the assistance from GIS staff from the Cape Cod Commission (CCC).

Figure IV-1 and Figure IV-2 show the land uses within the Barnstable Great Marsh estuary watershed. Land uses in the study area are grouped into 12 land use categories: 1) residential, 2) commercial, 3) industrial, 4) agricultural, 5) mixed use, 6) undeveloped, 7) public service/government, including road rights-of-way, 8) open space, 9) forest, 10) recreational, 11) freshwater, and 12) properties without assessor's land use codes. These land use categories are generally aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MassDOR, 2015). "Public service" in the MassDOR system is tax-exempt properties, including lands owned by government (*e.g.*, wellfields, schools, open space, roads) and private groups like churches and colleges.

Public service land uses are the dominant land use type in the overall Barnstable Great Marsh watershed and occupy 56% of the watershed area (Figure IV-3). Examples of these land uses are lands owned by town and state government (including golf courses, landfills, conservation lands, and wellhead protection lands), housing authorities, and churches. Residential land uses occupy the second largest area with 29% of the overall watershed area. The majority of the public service lands in the Barnstable Great Marsh watershed are wetlands surrounding the estuary, but these lands also include Sandy Neck Park, which extends along the barrier beach north of the main portion of the estuary. Other large public service lands within the watershed include: the West Barnstable Conservation Area, Marstons Mills Airfield, Olde Barnstable Fairgrounds Golf Course, Town of Barnstable Conservation lands around the Hathaway Ponds, Hyannis Golf Course, Cape Cod Community College, the Boy Scouts' Camp Greenough, the Town of Yarmouth Callery-Darling Conservation Area, and Dennis Highlands Golf Course.

Although the majority of the watershed area is public service land uses, the dominant parcel type within the whole watershed and the various subwatersheds are residential land uses. Residential parcels are 69% of the total parcel count in the overall West Barnstable subwatershed, 65% of parcels in the Mid-basin subwatershed, 73% of the BarnYarm subwatershed, 85% of the Bass Hole/Chase Garden Creek subwatershed, and 75% of all parcels in the overall Barnstable Great Marsh system watershed. Single-family residences (MassDOR land use code 101) are the dominant type of residential parcel; these represent 94% of all residential parcels in overall Barnstable Great Marsh system watershed and 90% of the residential parcel area in the overall watershed. It is notable that land classified by the town assessor as undeveloped is 9% of the overall watershed area.

In order to estimate wastewater flows within the Barnstable Great Marsh study area, MEP staff also obtained parcel-by-parcel water use data from the Sandwich Water District, Barnstable Fire District, Dennis Water Department, and Yarmouth Water Department for respective service areas within the watershed. A minimum of three years of water use was obtained with the following respective time periods: 2007 to 2010, 2006 to 2008, 2007 to 2009, and 2005-2011. The water use data was linked to the town parcel database and assessor's data by CCC GIS staff with QA/QC by MEP staff.

Measured water use is used to estimate wastewater-based nitrogen loading from individual parcels; average water use is used for each parcel with multiple years of data. The final wastewater nitrogen load for each parcel is based upon the measured water-use, wastewater nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system (see Section IV.1.2). All parcels are assumed to use on-site septic systems unless additional information is available.

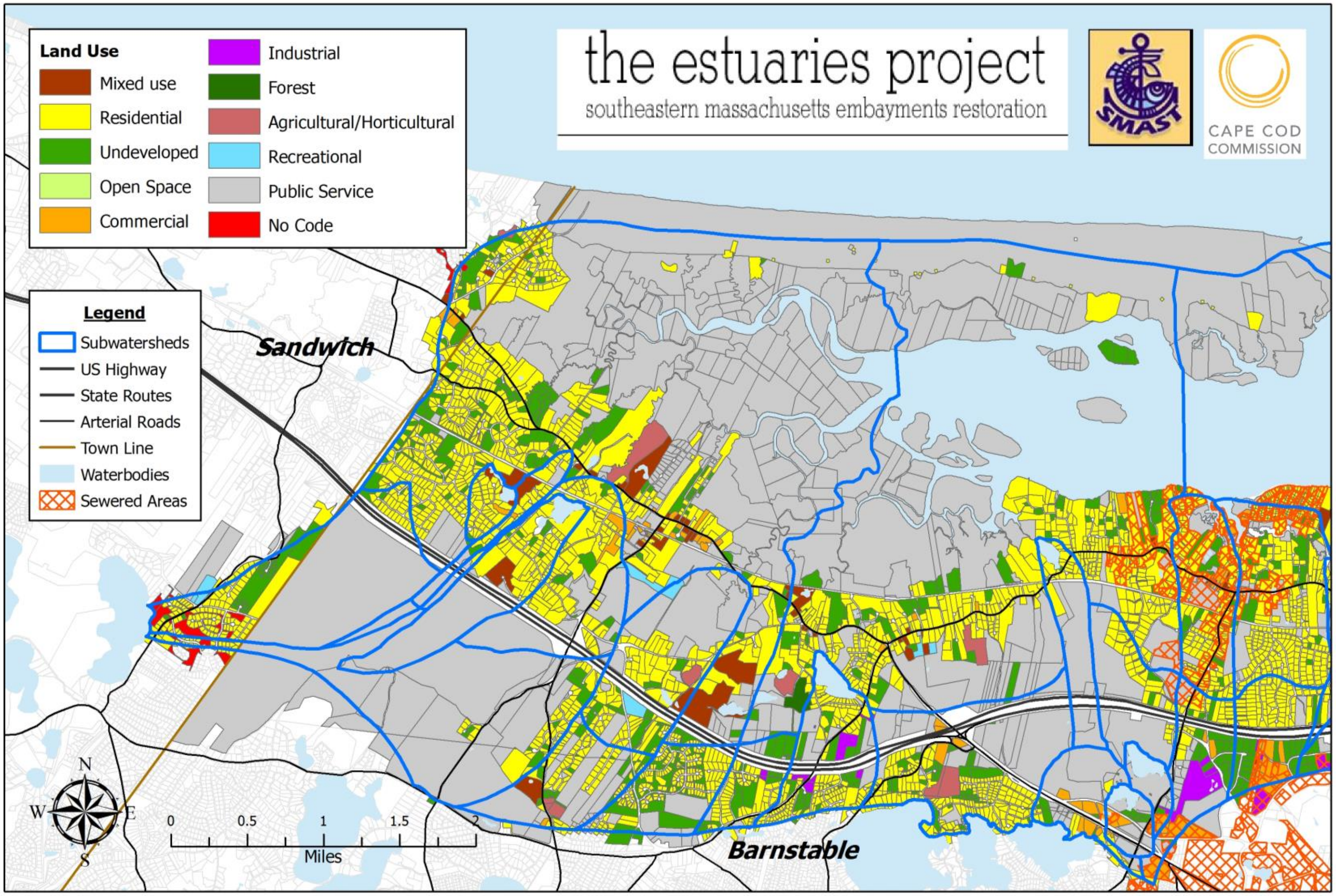


Figure IV-1. Land-use in the western half of the Barnstable Great Marsh system watershed and subwatersheds. Overall watershed includes portions of the towns of Barnstable, Yarmouth, Dennis, and Sandwich. Land use classifications are based on town assessor classifications and MADOR (2015) categories. Parcels along watershed boundaries are assigned to subwatersheds to 1) minimize the splitting of properties for future management purposes and 2) achieve a match of area with the modeled watersheds of 2% or less.

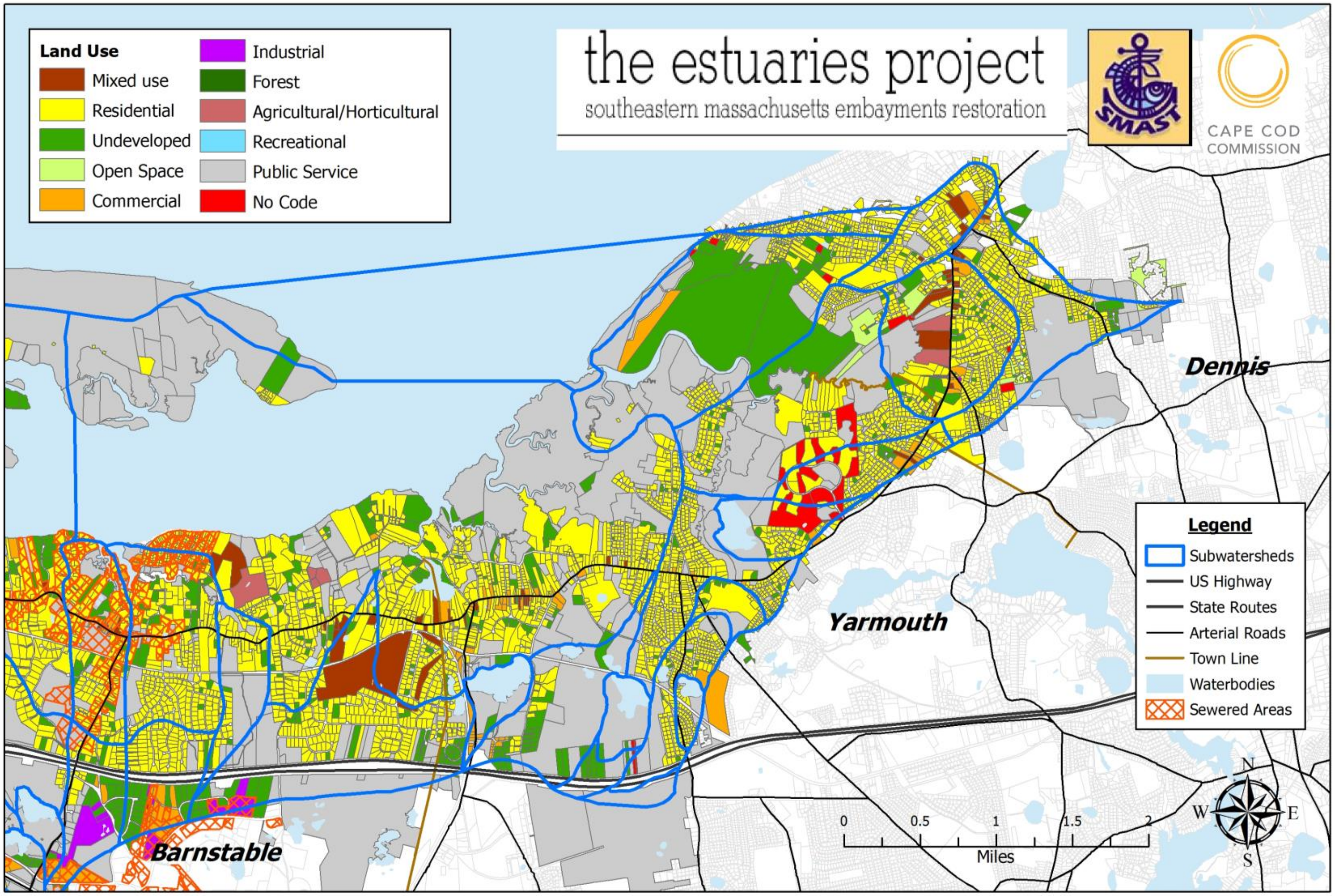


Figure IV-2. Land-use in the eastern portion of the Barnstable Great Marsh watershed and subwatersheds. Overall watershed includes portions of the towns of Barnstable, Yarmouth, Dennis, and Sandwich. Land use classifications are based on town assessor classifications and MADOR (2015) categories. Parcels along watershed boundaries are assigned to subwatersheds to 1) minimize the splitting of properties for future management purposes and 2) achieve a match of area with the modeled watersheds of 2% or less.

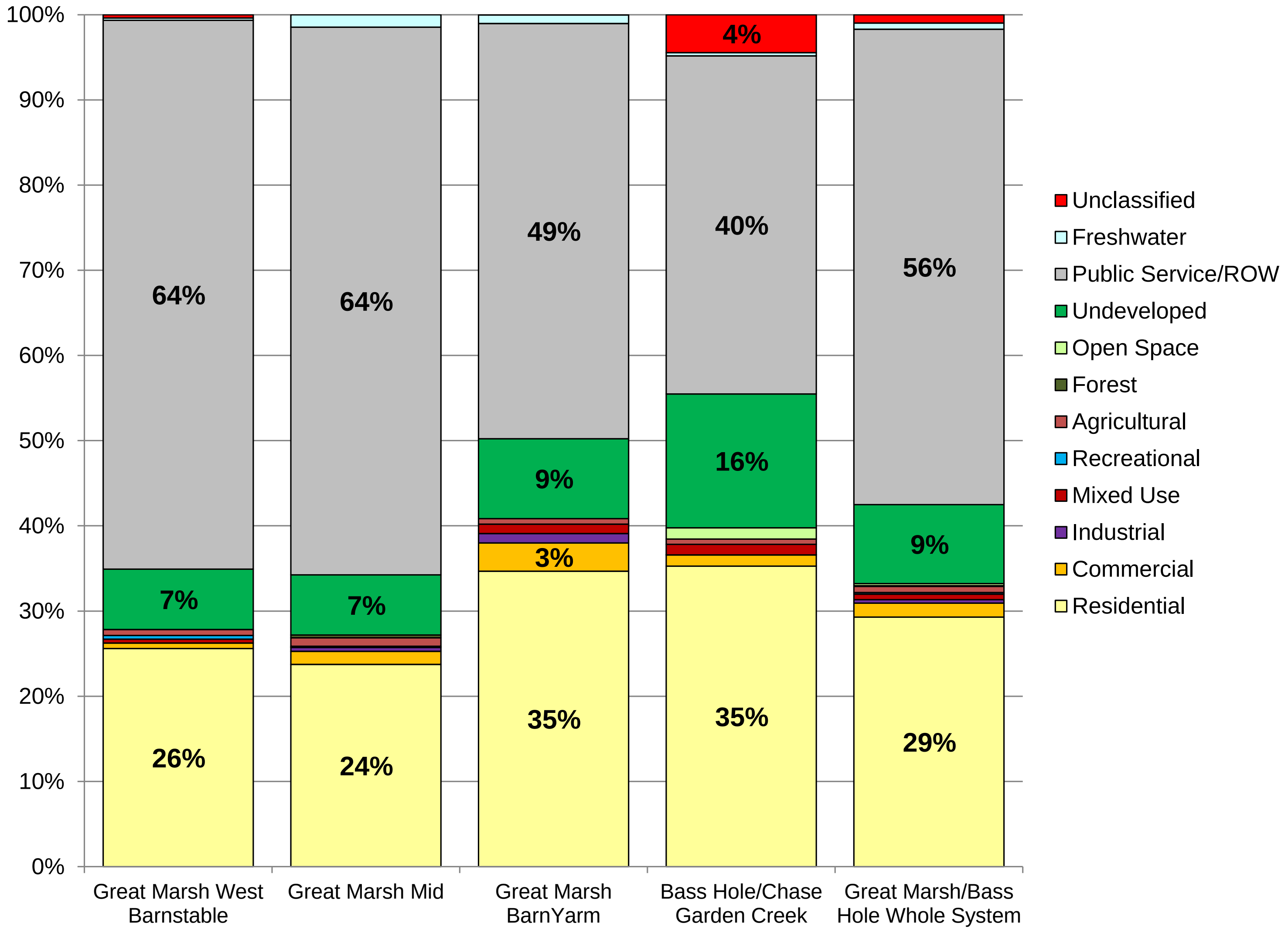


Figure IV-3. Distribution of land-uses by area within the Barnstable Great Marsh system watershed and selected component subwatersheds. Land use categories are generally based on town assessor's land use classification and grouping recommended by MADOR (2015). Unclassified parcels do not have an assigned land use code in the town assessor's databases. Only category percentages greater than or equal to 3% are labelled.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

The Massachusetts Estuaries Project septic system nitrogen loading rate is fundamentally based upon a *per capita* nitrogen load to the receiving aquatic system. Specifically, the MEP septic system wastewater nitrogen loading is based upon a number of studies and additional information that directly measured septic system and per capita loads on Cape Cod or in similar geologic settings (Nelson *et al*. 1990, Weiskel & Howes 1991, 1992, Koppelman 1978, Frimpter *et al.* 1990, Brawley *et al.* 2000, Howes, *et al.*, 2001, Costa *et al.* 2001). Variation in *per capita* nitrogen load has been found to be relatively small, with average annual *per capita* nitrogen loads generally between 1.9 to 2.3 kg person-yr^{-1}.

However, given the seasonal shifts in occupancy and rapid population growth throughout southeastern Massachusetts, decennial census data yields accurate estimates of total population only in selected watersheds. To correct for this uncertainty and more accurately assess current nitrogen loads, the MEP employs a water-use approach. The water-use approach is generally applied on a parcel-by-parcel basis within a watershed, where annual water meter data is linked to assessor's parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (*e.g.,* irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load that reaches the aquatic receptors downgradient in the aquifer.

All nitrogen losses within septic systems are incorporated into the MEP analysis. For example, information developed at the Massachusetts Alternative Septic System Test Center at the Massachusetts Military Reservation on Title 5 septic systems have shown nitrogen removals between 21% and 25%. Multi-year monitoring from the Test Center has revealed that nitrogen removal within the septic tank was small (1% to 3%), with most (20 to 22%) of the removal occurring within five feet of the soil adsorption system (Costa *et al.,* 2001). Downgradient studies of septic system plumes in similar soils indicate that further nitrogen loss during aquifer transport is negligible (Robertson *et al*., 1991, DeSimone and Howes 1996).

In its application of the water-use approach to septic system nitrogen loads, MEP staff has ascertained for the Estuaries Project region that while the *per capita* septic load is well constrained by direct studies, the consumptive use and nitrogen concentration data are less certain. As a result, MEP staff has derived a combined term for an effective N Loading Coefficient (consumptive use times N concentration) of 23.63, to convert water (per volume) to nitrogen load (N mass). This coefficient uses a *per capita* nitrogen load of 2.1 kg N person-yr^{-1} and is based upon direct measurements and corrects for changes in concentration that result from *per capita* shifts in water-use (*e.g.*, due to installing low plumbing fixtures or high versus low irrigation usage).

The nitrogen loads developed using this approach have been validated in a number of long and short term field studies where integrated measurements of nitrogen discharge from watersheds could be directly measured. Weiskel and Howes (1991, 1992) conducted a detailed watershed/stream tube study that monitored septic systems, leaching fields and the transport of the nitrogen in groundwater to adjacent Buttermilk Bay. This monitoring resulted in estimated annual *per capita* nitrogen loads of 2.17 kg (as published) to 2.04 kg (if new attenuation information is included). Further, modeled and measured nitrogen loads were determined for a small sub-watershed to Mashapaquit Creek in West Falmouth Harbor (Smith and Howes, manuscript in review) where measured nitrogen discharge from the aquifer was within 5% of the modeled N load. Another evaluation was conducted by surveying nitrogen discharge to the Mashpee River in reaches with swept sand channels and in winter when nitrogen attenuation is minimal. The modeled and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year in this type of ecological situation (Samimy and Howes, unpublished data).

While census-based population data has limitations in the highly seasonal MEP region, part of the regular MEP analysis is to compare expected water used based on average residential occupancy to measured average water uses. This is performed as a quality assurance check to increase certainty in the final results. This comparison has shown that the larger the watershed the better the match between average water use and occupancy. For example, in the cases of the combined Great Pond, Green Pond and Bournes Pond watershed in the Town of Falmouth and the Popponesset Bay/Eastern Waquoit Bay watershed, both of which cover large areas and have significant year-round populations, the septic nitrogen loading based upon the census data is within 5% of that from the water use approach. This comparison matches some of the variability seen in census data itself. Census blocks, which are generally smaller areas of any given town, have shown up to a 13% difference in average occupancy from town-wide occupancy rates. These analyses provide additional support for the use of the water use approach in the MEP study region.

Overall, the MEP water use approach for determining septic system nitrogen loads has been both calibrated and validated in a variety of watershed settings. The approach: (a) is consistent with a suite of studies on *per capita* nitrogen loads from septic systems in sandy soils and outwash aquifers; (b) has been validated in studies of the MEP Watershed "Module", where there has been excellent agreement between the nitrogen load predicted and that observed in direct field measurements corrected with other MEP Nitrogen Loading Coefficients (*e.g.*, stormwater, lawn fertilization); (c) the MEP septic nitrogen loading coefficient agrees with specific studies of consumptive water use and nitrogen attenuation between the septic tank and the discharge site; and (d) the watershed module provides estimates of nitrogen attenuation by freshwater systems that are consistent with a variety of ecological studies. It should be noted that while points b-d support the use of the MEP Septic N Coefficient, they were not used in its development. The MEP Technical Team has developed the septic system nitrogen load over many years, and the general agreement among the number of supporting studies has greatly enhanced the certainty of this critical watershed nitrogen loading term.

The independent validation of the water quality model (Section VI) and the reasonableness of the freshwater attenuation (Section IV.2) add additional weight to the nitrogen loading coefficients used in the MEP analyses and a variety of other MEP embayments. While the MEP septic system nitrogen load is the best estimate possible, to the extent that it may underestimate the nitrogen load from this source reaching receiving waters provides a safety factor relative to other higher loads that are generally used for septic systems in regulatory situations. The lower concentration results in slightly higher amounts of nitrogen mitigation (estimated at 1% to 5%)) needed to lower embayment nitrogen levels to a nitrogen target (*e.g.*, nitrogen threshold, cf. Section VIII). The additional nitrogen removal is not proportional to the septic system nitrogen level, but is related to the how the septic system nitrogen mass compares to the nitrogen loads from all other sources that reach the estuary (i.e. attenuated loads).

In order to estimate wastewater flows, MEP staff generally work with municipal or water supplier partners in the study watershed to obtain parcel-by-parcel water use information. In the Barnstable Great Marsh watershed, a minimum of three years of water use was obtained for each parcel with a municipal water connection from the various water suppliers in each of the towns. Based on this data, average daily water use for single-family residences with municipal water accounts in the Barnstable Great Marsh MEP study area is 216 gpd. If the Barnstable Great Marsh average flow is multiplied by 0.9 to account for consumptive use, the study area wastewater average flow for a single-family residence is 195 gpd. Water use is used as a proxy for wastewater generation from septic systems on all developed properties in the watershed. Wastewater-based nitrogen loading from the individual parcels using on-site septic systems is based upon the average water-use, nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system (see Section IV.1.2).

The Barnstable Great Marsh watershed is shared among the towns of Barnstable, Yarmouth, Dennis, and Sandwich. In order to provide a check on the measured water use, 2010 US Census average occupancies were reviewed. MEP staff reviewed US Census population values for the Towns of Barnstable, Yarmouth, and Dennis. Sandwich was not included because it occupies only a small portion of the watershed. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so for the purposes of Title 5 each person generates 55 gpd of wastewater. Based on data collected during the 2010 US Census, average occupancy within Barnstable, Yarmouth, and Dennis was 2.35, 2.12, and 2.05 people per housing unit. Multiplying these Census occupancies by the state Title 5 estimate of 55 gpd of wastewater *per capita* results in estimated water use per residence of 129, 116, and 113 gpd, respectively.

Given that such a high percentage of housing units on Cape Cod are occupied only on a seasonal basis, estimates of water use based on US Census data need to also include an adjustment for a seasonal population increase. Estimates of summer populations on Cape Cod and the Islands derived from a number of approaches (*e.g.*, traffic counts, garbage generation, and WWTF flows) generally suggest average summer population increases from two to four times the year-round residential populations measured during the US Census. For example, the Town of Barnstable Comprehensive Annual Financial Report estimated that the summer population was 2.9X the year-round population (Town of Barnstable, 2014). If it is assumed that seasonally-classified residential properties in Barnstable, Yarmouth, and Dennis are occupied at three times the 2010 year-round occupancy for three months, the estimated parcel water uses would be 194, 175, and 169 gpd, respectively. If the multiplier is increased to four times the year-round occupancy, the respective estimated water uses are 226, 204, and 197 gpd. If the "summer" period is increased from three months to four months to accommodate the so-called "shoulder seasons" and 3X year-round occupancy is used, the respective town water use estimates are 215 gpd, 194 gpd, and 188 gpd. This analysis of US Census data incorporating reasonable estimates of seasonal population increases suggests that average measured water use within the watershed is an appropriate basis of estimating water uses within the Barnstable Great Marsh watershed and is a suitable estimate for any parcels with private wells or for any future development additions. This analysis further suggests that population and water use information are in reasonable agreement and that the average water use is reasonably reflective of average wastewater estimates.

At the outset of the MEP, project staff decided to utilize the water use approach for determining residential wastewater generation by septic systems because of the inherent difficulty in accurately gauging actual occupancy in areas impacted by seasonal population fluctuations such as most of Cape Cod. The above analysis underscores some of the difficulty in using census information and strongly supports that measured water use, on average, provides the best estimate of wastewater generation within the study area.

Water use information exists for 74% of the 7,242 developed parcels within the Barnstable Great Marsh watershed. Parcels without water use accounts are assumed to utilize private wells for drinking water. These are properties that were classified with land use codes that indicate that they are developed (*e.g.*, 101 or 325) and have been confirmed as having buildings on them through a review of aerial photographs, but do not have a listed account in the water use databases. Of the 1,854 developed parcels without water use accounts, 1,534 (83%) are classified as single-family residences (land use code 101). These parcels are predominantly located within West Barnstable where municipal water supply is not available. All these parcels are assigned the MEP Barnstable Great Marsh watershed average water use of 216 gpd in the watershed nitrogen loading modules.

Wastewater Treatment Facilities and Alternative Septic Systems

When developing watershed nitrogen loading information, MEP project staff typically seeks additional information on enhanced wastewater treatment in the project study area. This information is reviewed and if judged reliable is included in the watershed nitrogen loading model.

MEP staff reviewed MassDEP Groundwater Discharge Permits (GWDPs) database and confirmed with MassDEP staff that the Kings Way wastewater treatment facility (WWTF) in Yarmouth is the only GWDP within the Barnstable Great Marsh watershed (personal communication, Brian Dudley, MassDEP, 9/15). A GWDP is required under MassDEP regulations for wastewater treatment systems with design flows greater than 10,000 gallons per day. The Kings Way WWTF treats wastewater from the Kings Way condominium and golf course complex north of Route 6A. The effluent discharge beds are located under the northern portions of the golf course and within the Chase Garden Creek Salt W LT10 subwatershed (subwatershed #40). MassDEP provided monthly discharge flows and effluent total nitrogen concentrations from 2010 through 2014 (**Figure IV-4**). Based on this information, MEP staff calculated the WWTF had an average effluent discharge of 45,750 gpd and an average annual nitrogen load of 274 kg.

While there is only one wastewater treatment facility discharging within the Barnstable Great Marsh watershed, the watershed also includes an area connected via sewers to the Hyannis Water Pollution Control Facility (HWPCF). This area is predominantly concentrated around Barnstable Village and includes portions of the Millway and Huckins Neck subwatersheds, as well as parcels in the watershed in Independence Park and along Route 132 (see Figures IV-1 and IV-2). Wastewater is collected from these parcels and transported to the HWPCF where it is treated and discharged. Since the wastewater nitrogen loads from these parcels is transported and discharged outside of the watershed, these parcels have no wastewater nitrogen load in the watershed nitrogen loading model.

MEP staff received a list of alternative, denitrifying septic systems in Barnstable, Dennis, and Yarmouth, as well as their total nitrogen effluent monitoring data, from the Barnstable County Department of Health and the Environment (personal communication, Brian Baumgaertel, 9/15). Sandwich is not part of the BCDHE tracking system. From the BCDHE database, project staff identified 23 potential denitrifying septic systems within the Barnstable Great Marsh watershed. After matching these to the parcel database and reviewing the available data, 20 of these had three or more sampling runs and were located on parcels with matching addresses; six are in Barnstable, two are in Yarmouth, and 12 are in Dennis. Four of the systems are on properties with private wells. The average TN concentrations of treated effluent for all these systems are included in the watershed nitrogen loading model.

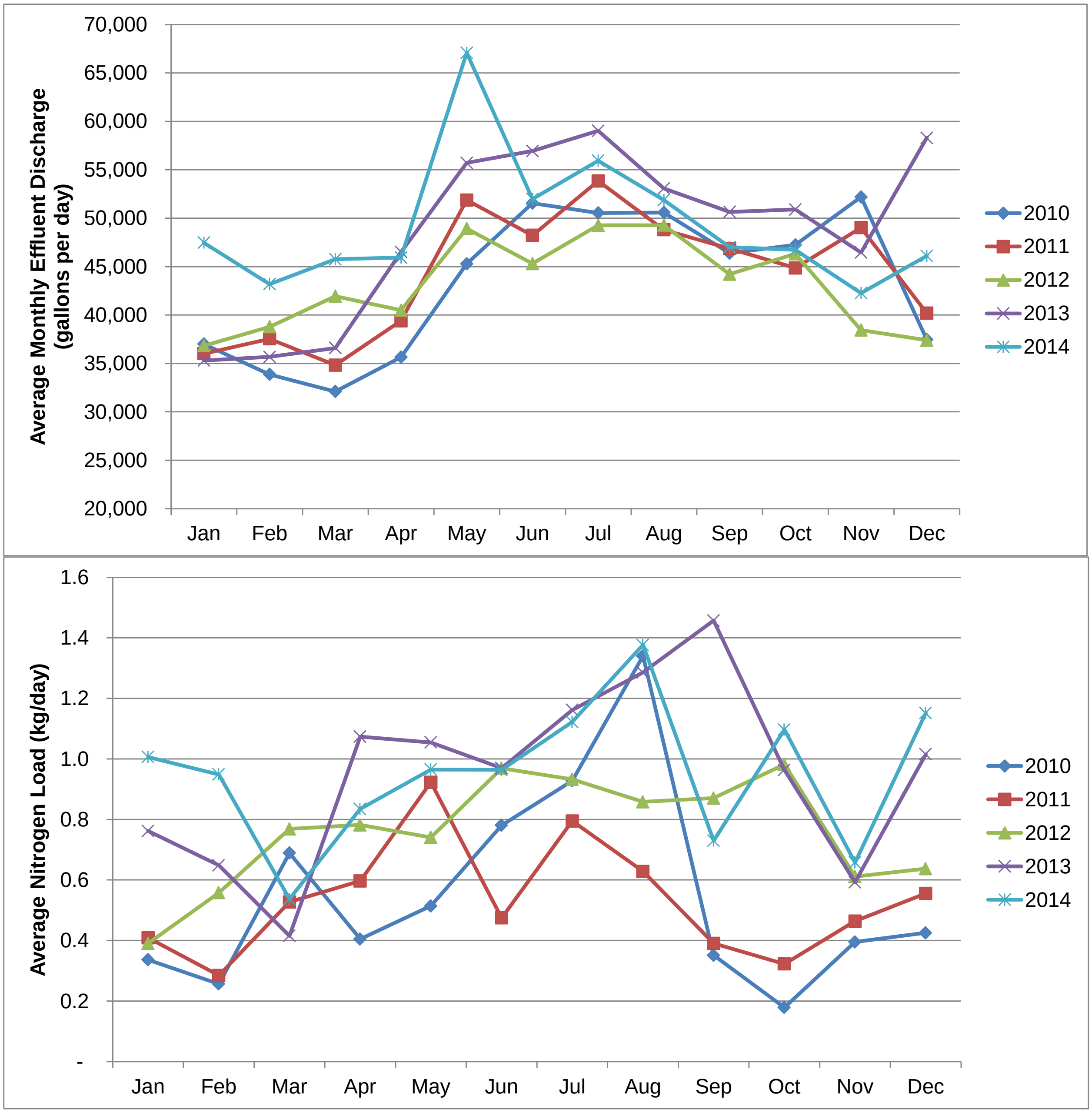


Figure IV-4. Average Monthly Effluent Discharge and Nitrogen Load at the Kings Way WWTF. The Kings Way WWTF is the only large wastewater treatment facility within the Barnstable Great Marshes watershed. The WWTF discharges within the Chase Garden Creek Salt W LT10 subwatershed (subwatershed #40). Based on 2010 to 2014 data supplied by MassDEP, the average effluent flow from the WWTF was 45,750 gpd and an average annual nitrogen load of 274 kg.

Nitrogen Loading Input Factors: Fertilized Areas, Golf Courses, and Agriculture

The second largest source of watershed nitrogen loading to estuaries is usually fertilized areas: lawns, golf courses, and cranberry bogs. Residential lawns are usually the predominant source within this category. In order to add this source to the watershed nitrogen loading model for the Barnstable Great Marsh Estuary, MEP staff reviewed available regional information about residential lawn fertilizing practices and incorporated site-specific information for cranberry bogs and agricultural areas in the watershed. Cranberry bog nitrogen loading was determined based on previous studies conducted in southeastern Massachusetts.

Residential lawn fertilizer use has rarely been directly measured in watershed-based nitrogen loading investigations. Instead, lawn fertilizer nitrogen loads have been estimated based upon a number of assumptions: a) each household applies fertilizer, b) cumulative annual applications are 3 pounds per 1,000 sq. ft., c) each lawn is 5000 sq. ft., and d) only 25% of the nitrogen applied reaches the groundwater (leaching rate). Because many of these assumptions had not been rigorously reviewed prior to the MEP, the MEP Technical Staff undertook an assessment of lawn fertilizer application rates and a review of leaching rates for inclusion in the Watershed Nitrogen Loading Sub-Model.

The initial effort in this assessment was to determine nitrogen fertilization rates for residential lawns in the Towns of Falmouth, Mashpee and Barnstable. This assessment, which was completed prior to the start of the MEP, accounted for proximity to fresh ponds and embayments. Based upon ~300 interviews and over 2,000 site surveys, a number of findings emerged: 1) average residential lawn area is ~5000 sq. ft., 2) half of the residences did not apply lawn fertilizer, and 3) the weighted average application rate was 1.44 applications per year, rather than the 4 applications per year recommended on the fertilizer bags. Integrating the average residential fertilizer application rate with a nitrogen leaching rate of 20% results in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn; these factors are used in the MEP nitrogen loading calculations. It is likely that this still represents a conservative estimate of nitrogen load from residential lawns. It should be noted that professionally maintained lawns in the three town survey were found to have the higher rate of fertilizer application and hence higher estimated annual contribution to groundwater of 3 lb/yr. It should be noted that a recent review of fertilizer leaching rates for Cape Cod watersheds indicated that the MEP leaching rate of 20% is fully supported from analysis of all appropriate data, and was a reasonable value for watershed nitrogen modeling in watersheds with soils similar to those on Cape Cod.

In addition to residential fertilizer nitrogen within the watershed, there is also fertilizer nitrogen from portions of six golf courses. A number of the golf courses are shared with other MEP watersheds; in these cases, MEP staff utilized previous course- and turf-specific fertilizer application information obtained from course superintendents. For the “new” courses, MEP staff tried a number of times to contact course superintendents in order to obtain similar information. Staff had previously obtained nitrogen fertilizer information from:

a) Olde Barnstable Fairgrounds Golf Course (personal communication, Bruce McIntryre, Superintendent, 5/06; Centerville River MEP report (Howes, *et al*., 2006a));
b) Bayberry Hills Golf Course (personal communication, Rick Lawlor, Superintendent, 2/09; Bass River MEP report (Howes, *et al*., 2011)); and
c) Dennis Highlands Golf Course (personal communication, Mike Cummings, Superintendent, 6/10; Bass River MEP report (Howes, *et al*., 2011)).

MEP staff also obtained nitrogen fertilizer information from the Cummaquid Golf Club (personal communication, Dana Hancock, Superintendent, 8/15), but did not obtain information from the Golf Club at Yarmouthport or the Hyannis Golf Course. Since course- and turf-specific fertilizer information was not obtained for these two courses, nitrogen application rates and loads were determined based on average application rates developed from 23 courses within the MEP region that have provided this information during previous MEP watershed nitrogen loading assessments. Based on these regional average application rates, average turf application rates for Yarmouthport and Hyannis Golf Courses were estimated to be: greens, 3.5; tees, 3.5; fairways, 3.2, and rough, 2.4 (in lbs/1,000 sq. ft. per year).

In order to develop nitrogen loads for each of the golf courses within the Barnstable Great Marsh watershed, MEP staff reviewed the layout of each golf course from aerial photographs, classified the various turf types (*i.e.*, greens, tees, fairways, and roughs) and determined which subwatershed in which the various turf types were located. The course-specific fertilizer application rates were then combined with a standard MEP 20% turf nitrogen leaching rate and subwatershed-specific nitrogen loads were developed.

Nitrogen loads were also added for site-specific agricultural land uses. Cranberry bog fertilizer application rate and percent nitrogen attenuation are based on an enhanced review of nitrogen export from cranberry bogs in southeastern Massachusetts (DeMoranville and Howes, 2009; Howes and Teal, 1995). This review found that nitrogen export from cranberry bogs differs depending on whether water continuously flows through the bog or is pumped or diverted onto the bog (non-flow through bogs) from an outside source of water. Based on this review, MEP analyses use annual nitrogen exports of 6.95 kg/ha for non-flow through bogs and 23.08 kg/ha for flow through bogs. MEP staff reviewed the configuration of the 11 bogs within the Barnstable Great Marsh watershed and assigned all bogs the flow through bog nitrogen loading rate. In addition, one of the bogs was not assigned a nitrogen loading addition because it discharges into Lake Wequaquet. The areas of the bogs are based on a MassDEP GIS coverage that is maintained by MassDEP for Water Management Act permitting (personal communication, Jim McLaughlin, MassDEP SERO, 1/13).

Nitrogen loads were also added based on agricultural animals within the watershed. MEP staff received a listing of registered horse stables in the Town of Barnstable (personal communication, Sarah Crocker, Barnstable Health Department, 12/15) and reviewed farm activity information available through internet searches and review of aerial photographs in order to develop estimates of farm animal counts. Because of the uncertainties in the development of these estimates, it is assumed that these counts and the resulting agricultural animal loads are somewhat conservative. Species-specific nitrogen loads were developed based on USDA and other species-specific research on nitrogen manure characteristics, including leaching to groundwater. Loads were assigned to individual farm lots based on the animal counts. Details of these loads are included in the MEP Data Disk that accompanies this report.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas in the Barnstable Great Marsh assessment are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes, *et al*., 2001). The factors are similar to those utilized by the CCC's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and MassDEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III).

Factors used in the MEP nitrogen loading analysis for the Barnstable Great Marsh watershed are summarized in Table IV-2.

Road and building areas are based on GIS coverages. The road areas are based on GIS information developed by the Massachusetts Executive Office of Transportation, which provides road, sidewalk, and road shoulder widths for various road segments (April 2012 GIS coverage available through MassGIS). Building footprint areas are based on MassGIS coverage developed based on digitized footprints from 2011 to 2012 aerial orthophotographs supplemented with LiDAR data. MEP staff utilized GIS techniques to sum both of these sets of areas by subwatershed within the Barnstable Great Marshes watershed in order to determine nitrogen loads from these impervious surfaces. Project staff also checked the road information against parcel-based rights-of-way.

Table IV-2. Primary Nitrogen Loading Factors used in the Barnstable Great Marsh MEP analyses. General factors are from MEP modeling evaluation (Howes, *et al*., 2001). Site-specific factors are derived from watershed-specific data.

Nitrogen Concentrations:	**mg/l**
Road Run-off	1.5
Roof Run-off	0.75
Natural Area Recharge	0.072
Direct Precipitation on Embayments and Ponds	1.09
Wastewater Coefficient	23.63
Fertilizers:	
Average Residential Lawn Size (sq ft)[1]	5,000
Residential Watershed Nitrogen Rate (lbs/lawn)[1]	1.08
Leaching rate	20%
Cranberry Bogs nitrogen release – flow through bogs (kg/ha/yr)	23.08
Cranberry Bogs nitrogen release – pump on/pump off bogs (kg/ha/yr)	6.95
Nitrogen Loading Rates for farm animals were based on loads determined in other MEP assessments and USDA guidance, while fertilizer loading rates for golf courses were generally based on course-specific use.	

Recharge Rates:	**in/yr**
Impervious Surfaces	40
Natural and Lawn Areas	27.25
Water Use/Wastewater:	
Existing developed single-family residential parcels wo/water accounts and buildout residential parcels:	216 gpd[2]
Existing developed parcels w/water accounts:	Measured annual water use
Commercial and Industrial Buildings without/WU and buildout additions[3]	
Commercial	
Wastewater flow (gpd/1,000 ft2 of building):	95
Building coverage:	10%
Industrial	
Wastewater flow (gpd/1,000 ft2 of building):	39
Building coverage:	5%
Average Single Family Residence Building Size[3] (sq ft)	2,008

Notes:
1) Data from MEP lawn study in Falmouth, Mashpee & Barnstable of over 2,000 lawns (2001).
2) Based on average measured flow in the MEP Barnstable Great Marsh watershed.
3) Based on characteristics of similarly classified properties with the MEP Barnstable Great Marsh watershed.

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information is linked to the parcel coverages, parcels are assigned to various watersheds based initially on whether at least 50% or more of the land area of each parcel is located within a respective subwatershed. Following the assigning of boundary parcels, all large parcels are examined individually and are split (as appropriate) in order to obtain less than a 2% difference between the total land area of each subwatershed and the sum of the area of the parcels within each subwatershed. This effort results in “parcelized” watersheds that can be more easily used during the development of management strategies and subsequent regulatory discussions.

The review of individual parcels straddling watershed boundaries includes corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Building footprints areas, for example, are based on MassGIS review of aerial photographs and this is assigned based on individual parcels and subwatershed lines. Project staff used the average single-family residence building footprint based on available properties in the MEP study area (2,008 sq ft) for any similar residential units without footprint information (*e.g.*, buildings since the MassGIS coverage was developed). Individualized information for parcels with atypical nitrogen loading (condominiums, golf courses, etc.) is also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure generally have a negligible effect on the total nitrogen loading to the Barnstable Great Marsh estuary. The assignment effort is undertaken to better define sub-estuary loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

Following the assignment of all parcels, subwatershed modules were generated for each of the 45 subwatersheds in the Barnstable Great Marsh study area. These subwatershed modules summarize, among other things: water use, parcel area, frequency, and private wells by land use categories and road and fresh surface water areas within a given subwatershed. All relevant nitrogen loading data is assigned to each subwatershed. Individual sub-watershed information is then integrated to create the Barnstable Great Marsh Watershed Nitrogen Loading module with summaries for each of the individual 45 subwatersheds. The subwatersheds are generally paired with functional embayment/estuary units for the Linked Watershed-Embayment Model’s water quality component.

For management purposes, the aggregated estuary watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Barnstable Great Marsh overall watershed, the major types of nitrogen loads are: wastewater (*e.g.*, septic systems), fertilizers (including contributions from agriculture and golf courses), impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-3). The output of the watershed nitrogen-loading model is the annual mass (kilograms) of nitrogen added to the contributing area of each component sub-embayment, by each source category (Figure IV-5). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation in streams and ponds during transport to the estuarine system before use in the embayment water quality sub-model.

Table IV-3. Barnstable Great Marsh Watershed Nitrogen Loads. Nitrogen loads are listed by various sources and by subwatershed. Unattenuated nitrogen loads are a sum of all sources without including natural nitrogen attenuation in fresh surface waters. Attenuated nitrogen loads are based on measured and assigned attenuation factors for upgradient streams and freshwater ponds. Stream attenuation factors are based on measured loads (see Section IV.2). All nitrogen loads are kg N yr-1.

		Barnstable Great Marsh/Bass Hole N Loads by Input (kg/y):												Present N Loads			Buildout N Loads		
Watershed Name	Shed ID#	Wastewater	WWTF	Residential Fertilizers	Golf Course Fertilizers	Cranberry Bog Fertilizers	Crop Fertilizers	Farm Animals	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout	% of Pond Outflow	UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Whole System		53,984	274	3,549	2,792	373	123	1,525	5,247	1,903	3,842	16,326		73,613		66,221	89,939		81,196
Great Marsh West Barnstable		10,817	-	720	622	373	25	840	1,103	304	1,361	4,191		16,165		14,084	20,356		17,685
BarnHbr GT10W	1	1,095	-	75	-	-	-	-	125	-	125	311		1,419		1,419	1,731		1,731
BarnHbr LT10 West Barnstable	10	4,470	-	286	-	157	-	143	391	-	724	1,862		6,171		6,171	8,033		8,033
BarnHbr GT10 West Barnstable	11	1,208	-	82	-	-	-	-	62	-	40	101		1,392		1,392	1,493		1,493
Lawrence Pond	MEP	99	-	6	-	-	-	-	5	160	16	215	16%	286		102	502		193
Bog Pond	BP	286	-	20	-	5	-	-	24	26	7	79	78%	368		184	447		223
Alder Creek		322	-	21	-	84	-	13	27	8	21	124		496		288	619		361
Boat Cove Creek		1,939	-	137	350	10	10	347	304	110	299	900		3,506		3,214	4,405		4,025
Bridge Creek		1,398	-	93	272	116	15	337	165	-	130	599		2,526		1,314	3,125		1,625
Great Marsh Mid		8,649	-	572	606	-	-	-	965	813	860	2,639		12,465		10,060	15,103		12,398
BarnHbr GT10 MidW	13	560	-	32	-	-	-	-	49	-	28	70		668		668	738		738
BarnHbr GT10 - Exit 6	14	1,911	-	101	492	-	-	-	281	-	93	354		2,878		2,878	3,232		3,232
Barnstable Hbr GT10 MidE	19	219	-	15	-	-	-	-	44	-	49	31		327		327	358		358
Barnstable Hbr LT10 Mid	20	3,471	-	233	-	-	-	-	327	-	573	1,613		4,605		4,605	6,218		6,218
BFD Wells	BFD	233	-	5	43	-	-	-	26	69	10	65	24%	388		282	453		329
Hinckley Pond	HP	256	-	14	71	-	-	-	28	64	34	86	100%	466		233	552		276
Garretts Pond	GP	1,208	-	86	-	-	-	-	120	127	46	301	100%	1,587		793	1,887		944
Hathaway Pond N	HPN	61	-	0	-	-	-	-	24	100	11	63	100%	196		78	259		99
Lake Wequaquet - Main Basin	MEP/SMAST	557	-	68	-	-	-	-	50	283	12	42	15%	989	86%	142	1,031	86%	148
Bearses Pond	MEP/SMAST	143	-	13	-	-	-	-	11	71	3	10	24%	245	84%	39	256	84%	41
Shallow Pond	MEP/SMAST	29	-	3	-	-	-	-	5	99	2	4	28%	89	84%	14	93	84%	15
Great Marsh BarnYarm		14,847	-	1,028	580	-	98	685	1,599	505	976	7,294		20,318		19,379	27,612		26,539
BarnHbr GT10 Cummaquid	26	1,924	-	98	205	-	-	-	180	-	131	1392		2,538		2,538	3,930		3,930
BarnHbr GT10 - Yarmouthport	28	393	-	15	-	-	-	-	52	26	68	428		554		554	982		982
BarnHbr LT10 - BarnYarm	31	7,493	-	538	246	-	29	52	651	-	489	1872		9,500		9,500	11,372		11,372
BFD Wells	BFD	694	-	15	129	-	-	-	78	206	30	194	71%	1,152		839	1,346		977
Yarm_Well1	YW1	627	-	46	-	-	-	-	58	22	11	18	34%	764		681	782		700
Dennis Pond	DP	214	-	-	-	-	-	-	16	215	10	40	100%	455		227	495		248
Greenough Pond	GP	11	-	-	0	-	-	-	6	36	8	46	31%	61		30	107		54
Huckins Neck Total		743	-	96	-	-	-	13	210	-	88	2,500		1,149		1,149	3,650		3,650
Huckins Neck GT10	21	447	-	18	-	-	-	-	83	-	42	2,406		590		590	2,997		2,997
Huckins Neck LT10	22	296	-	77	-	-	-	13	127	-	46	94		559		559	653		653
Millway Total		2,748	-	220	-	-	68	619	350	-	141	802		4,145		3,860	4,948		4,628
Millway GT10	23	590	-	36	-	-	-	-	94	-	38	314		758		758	1,072		1,072
Millway LT10	24	1,595	-	152	-	-	-	-	211	-	70	325		2,027		2,027	2,352		2,352
Maraspin Creek		563	-	32	-	-	68	619	44	-	33	163	100%	1,360		1,074	1,523		1,203

Table IV-3 (continued). Barnstable Great Marsh Watershed Nitrogen Loads. Nitrogen loads are listed by various sources and by subwatershed. Unattenuated nitrogen loads are a sum of all sources without including natural nitrogen attenuation in fresh surface waters. Attenuated nitrogen loads are based on measured and assigned attenuation factors for upgradient streams and freshwater ponds. Stream attenuation factors are based on measured loads (see Section IV.2). All nitrogen loads are kg N yr-1.

		Barnstable Great Marsh/Bass Hole N Loads by Input (kg/y):												Present N Loads			Buildout N Loads		
Watershed Name	Shed ID#	Wastewater	WWTF	Residential Fertilizers	Golf Course Fertilizers	Cranberry Bog Fertilizers	Crop Fertilizers	Farm Animals	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout	% of Pond Outflow	UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Bass Hole/Chase Garden Creek		19,671	274	1,229	984	-	-	-	1,580	281	646	2,202		24,665		22,698	26,867		24,575
Barnstable Hbr LT10 Inlet	45	6	-	1	-	-	-	-	2	-	21	0		30		30	30		30
Chase Garden Creek Salt W		9,130	274	583	502	-	-	-	818	281	271	999		11,858		10,828	12,857		11,642
Chase Garden Crk Salt W GT10	38	1,389	-	70	162	-	-	-	162	-	24	92		1,807		1,807	1,898		1,898
Chase Garden Crk Salt W LT10	40	2,413	274	151	110	-	-	-	215	-	126	327		3,289		3,289	3,616		3,616
Yarm_Well1	YW1	1,142	-	84	-	-	-	-	105	40	20	34	62%	1,391		1,240	1,424		1,274
Whites Brook Salt		4,185	-	278	230	-	-	-	336	241	101	547		5,372		4,492	5,919		4,854
Whites Brook Salt GT10	32	41	-	-	86	-	-	-	7	-	7	0		141		141	141		141
Whites Brook Salt LT10	36	2,787	-	199	-	-	-	-	225	-	45	93		3,256		3,256	3,349		3,349
Matthews Pond	MP	604	-	41	143	-	-	-	58	159	19	54	100%	1,024		512	1,079		539
Greenough Pond	GP	23	-	-	0	-	-	-	13	79	17	101	69%	132		66	233		117
Whites Brook Fresh		731	-	39	-	-	-	-	33	3	13	298		818		516	1,116		707
Chase Garden Creek Salt E		8,402	-	459	483	-	-	-	561	-	169	1,071		10,073		9,136	11,144		10,067
Chase Garden Creek Salt E GT10	39	3,330	-	190	483	-	-	-	238	-	63	323		4,303		4,303	4,626		4,626
Chase Garden Crk Salt E LT10	41	3,736	-	179	-	-	-	-	220	-	74	515		4,209		4,209	4,724		4,724
Chase Garden Creek Fresh		1,335	-	91	-	-	-	-	102	-	33	233		1,561		624	1,794		718
Bass Hole		2,134	-	185	-	-	-	-	200	-	185	132		2,704		2,704	2,836		2,836
Bass Hole GT10	43	762	-	73	-	-	-	-	66	-	11	47		914		914	960		960
Bass Hole LT10	44	1,371	-	111	-	-	-	-	133	-	174	86		1,790		1,790	1,876		1,876

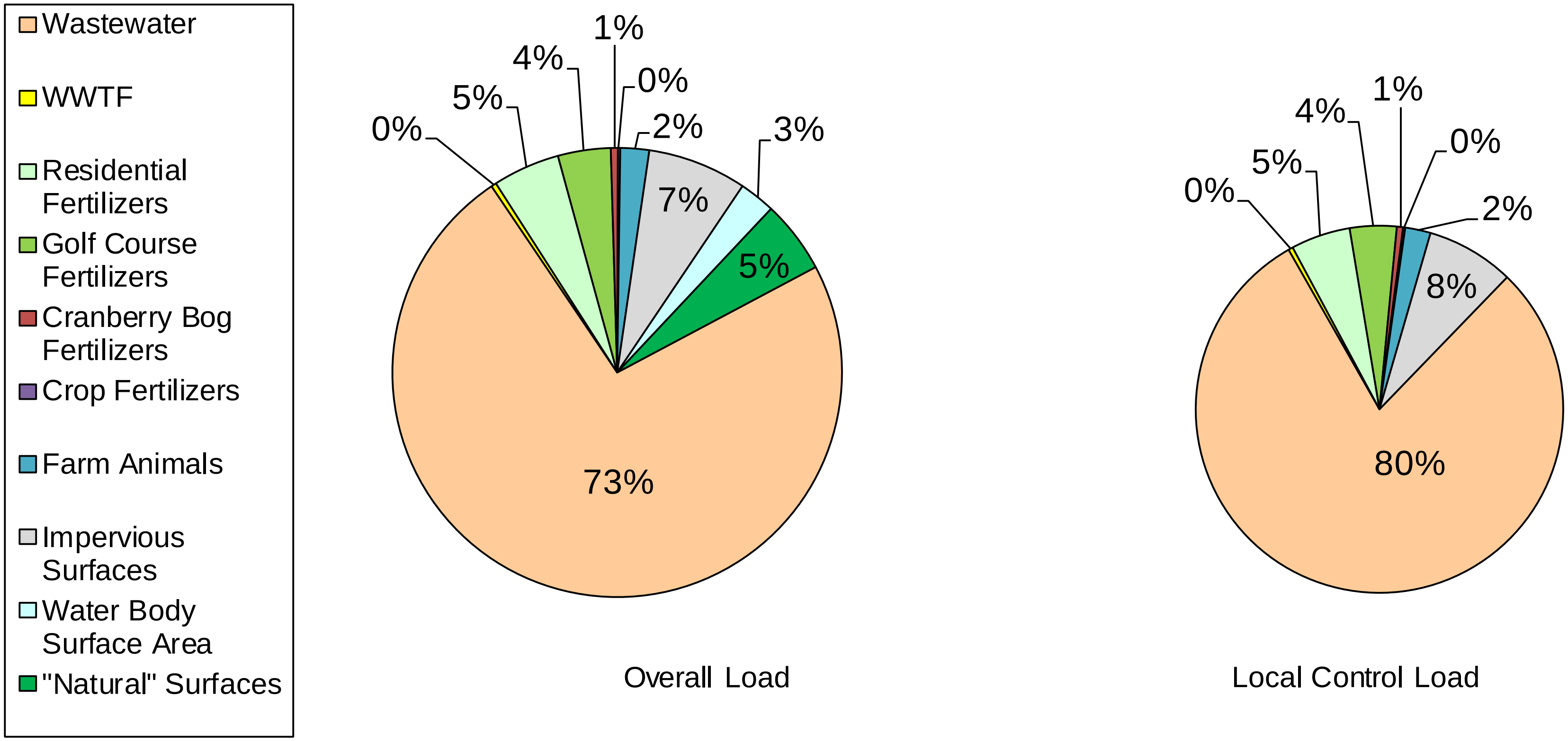
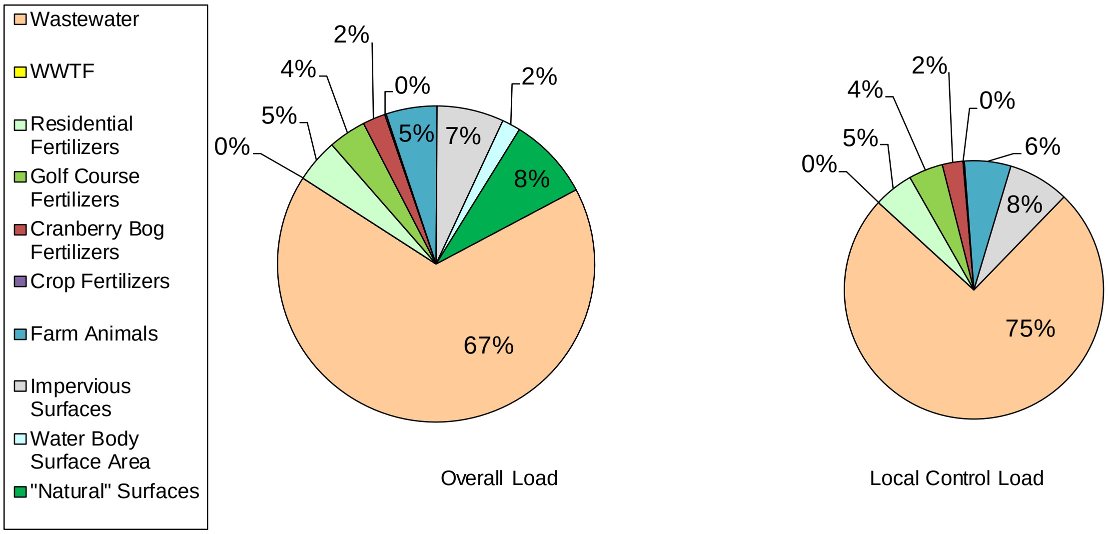
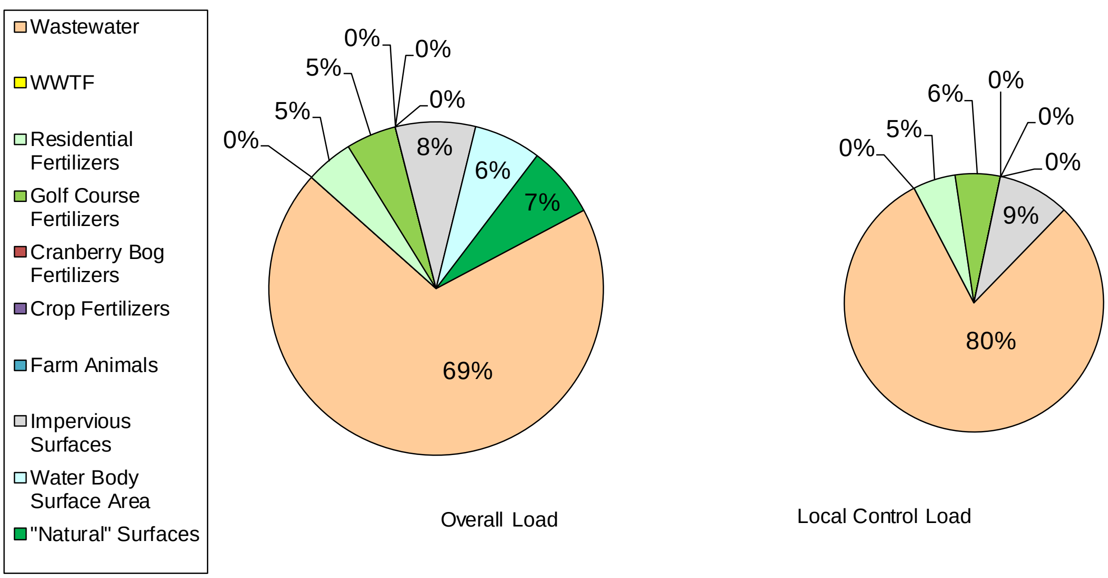


Figure IV-5a. Source-specific unattenuated watershed nitrogen loads (by percent) to the whole Barnstable Great Marsh watershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control. Watershed loads do not include nitrogen load on the surface of the estuary; this load is greater than all other categories of nitrogen load except wastewater.

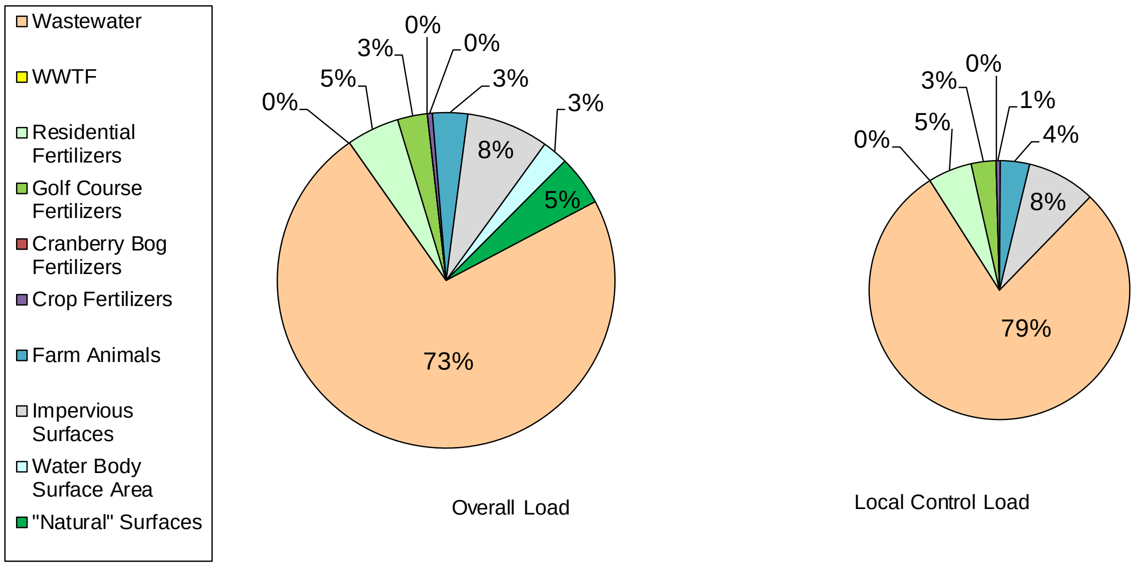


B. Great Marsh West Barnstable

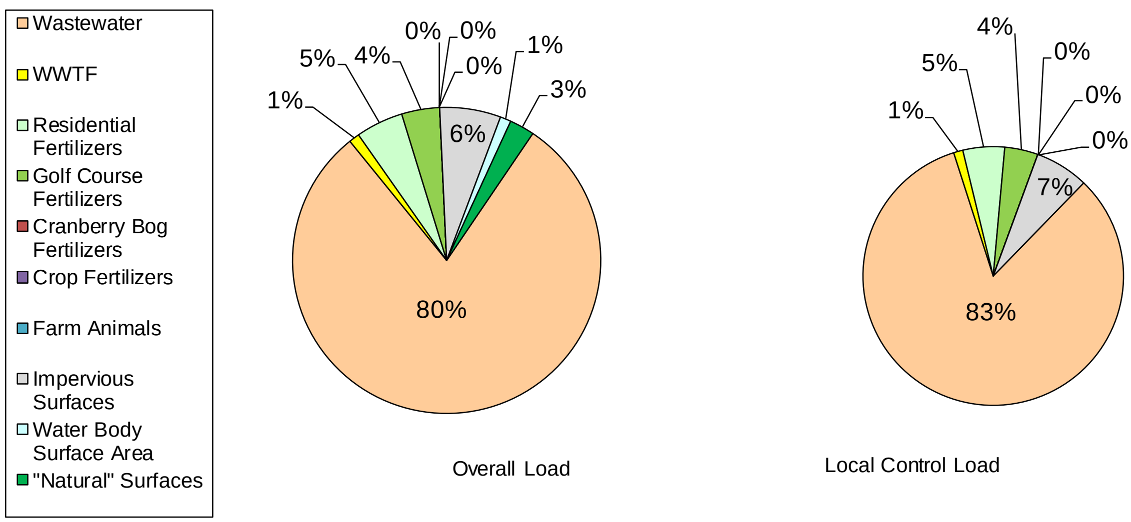


C. Great Marsh Mid

Figure IV-5b,c. Source-specific unattenuated watershed nitrogen loads (by percent) to the B) Great Marsh West Barnstable subwatershed and C) Great Marsh Mid subwatershed. “Overall Load” is the total nitrogen input within the watershed, while the “Local Control Load” represents only those nitrogen sources that could potentially be under local regulatory control. Watershed loads do not include nitrogen load on the surface of the estuary.



D. Great Marsh BarnYarm



E. Bass Hole/Chase Garden Creek

Figure IV-5d,e. Source-specific unattenuated watershed nitrogen loads (by percent) to the D) Great Marsh BarnYarm subwatershed and D) Bass Hole/Chase Garden Creek subwatershed. “Overall Load” is the total nitrogen input within the watershed, while the “Local Control Load” represents only those nitrogen sources that could potentially be under local regulatory control. Watershed loads do not include nitrogen load on the surface of the estuary.

Freshwater Pond Nitrogen Loads

Freshwater ponds are one of the primary watershed locations where natural nitrogen attenuation occurs and this attenuation is included in the MEP watershed nitrogen loading model. Freshwater ponds in aquifer systems like those of Cape Cod are generally kettle hole depressions of the land surface that intercept the surrounding groundwater table revealing what some call “windows on the aquifer.” Groundwater typically flows into the pond along the upgradient shoreline, then lake water flows back into the groundwater system along the downgradient shoreline. Occasionally, ponds will also have a stream outlet or herring run that also acts as a discharge point which sometimes has flow control to artificially manipulate water level and outflow rate. Ponds may also be connected to each other through streams and rivers, as well as connections that have been developed for cranberry bog operations.

Since watershed nitrogen loads flow into the ponds along with the groundwater, the pond biomass (plants and animals) have the opportunity to incorporate some of the nitrogen, as well as transporting/burying some of it to the pond sediments. As the nitrogen is captured and used in the pond ecosystem, it is also changed amongst its various oxidized and reduced forms. These interactions also allow for some chemical denitrification and release of some of the nitrogen to the atmosphere, as well as permanent burial in the pond sediments of some portion of the load that the pond receives. Through the cumulative effect of these interactions with the pond ecosystem, some of the nitrogen from the pond watershed is removed and is not transferred downgradient or downstream. If this reduced (or attenuated) load does not encounter any streams or other ponds, it will eventually discharge to the downgradient estuary. If it enters another pond or stream prior to discharge, this load can be further attenuated (see Section IV.2 for stream attenuation). In the nitrogen loading summary in Table IV-3, the unattenuated loads are those without any natural nitrogen attenuation included, while the attenuated loads include the attenuation within ponds, streams, and, in some cases, the cumulative effect of attenuation within a number of ponds and streams as the water moves toward discharge into the estuary.

Nitrogen attenuation in freshwater ponds has generally been found to be at least 50% in MEP analyses, so this value is generally used as a standard MEP default attenuation rate when sufficient pond-specific data is not available. Detailed studies of southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM, 2001) have supported a 50% attenuation factor as a reasonable, somewhat conservative rate. However, in some cases, if sufficient monitoring information is available, a pond-specific attenuation rate is incorporated into the watershed nitrogen loading model [*e.g.*, 87%, Mystic Lake; 40%, Middle Pond; and 52%, Hamblin Pond in the Three Bays MEP Report (Howes, *et al.*, 2006b) and Mashpee-Wakeby Pond, 86%, and Santuit Pond, 75%, in the Popponesset Bay MEP Report (Howes et al. 2004)]. In order to estimate nitrogen attenuation in the ponds, available physical and water quality data for each pond is reviewed. Available bathymetric information is reviewed relative to measured pond temperature profiles to determine whether an epilimnion (*i.e.*, well mixed, uniform temperature, upper portion of the water column) exists in each pond. This step is completed to assess whether available data is influenced significantly by nitrogen regeneration from the pond sediments. Bathymetric information is necessary to develop a residence or turnover time and complete an estimate of nitrogen attenuation. Collectively, a standard 50% nitrogen attenuation rate is assigned to ponds with delineated watersheds in MEP nitrogen loading models unless sufficient information is available regarding the physical structure of the pond and its water quality conditions to reasonably assign a different pond-specific rate.

In the Barnstable Great Marsh watershed, MEP staff reviewed available data sources for available monitoring and physical characterization data for the 10 ponds with delineated subwatersheds within the overall Great Marsh watershed, as well as the four ponds shared with other MEP watersheds (Table IV-4). Among the 10 ponds within the watershed, review of available bathymetric and available water quality data generally was insufficient to assign a pond-specific nitrogen attenuation rate other than the standard MEP 50% rate. A reasonable bathymetric map is a prerequisite to developing a pond-specific attenuation rate: Bog Pond, Hathaway Pond South, Hinckley Pond, and Mill Pond in Barnstable and Elishas Pond, Greenough Pond, and Matthews Pond in Yarmouth do not have bathymetric maps. Of the remaining three ponds with bathymetric maps, water quality data has been collected during the annual Cape Cod Pond and Lakes Stewardship (PALS) water quality snapshots, but data has not been collected during the rest of the summer to provide proper context for the PALS data and assign a pond-specific nitrogen attenuation rate other than the standard MEP 50% rate.

Hundreds of ponds on Cape Cod have been sampled through the regional Cape Cod Pond and Lake Stewards (PALS) Snapshots and the initiative of local volunteer pond sampling programs. The annual PALS Snapshots are regional volunteer, late-summer pond samplings supported for the last thirteen years by SMAST and the Cape Cod Commission, with pro bono laboratory services provided by the Coastal Systems Program Analytical Facility at SMAST. Sampling protocols developed through the PALS program (Eichner *et al*., 2003) have been used for more extensive pond sampling programs in many communities on Cape Cod. Sampling under these protocols has included field collection of temperature and dissolved oxygen profiles and sampling of standardized depths that include some evaluation of the impact of sediment nutrient regeneration. PALS water samples are analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH. In some cases, town programs have generated sufficient sampling data collected throughout a number of summers that modified MEP nitrogen attenuation rates can be reliably assigned to freshwater ponds.

PALS sampling was designed to test whether ponds were experiencing impaired conditions during the warmest part of the summer; if a given pond was experiencing impaired conditions during the PALS sampling, then more expansive and targeted sampling was recommended to assess the regular severity of those conditions. If conditions were satisfactory during numerous PALS snapshots, pond sampling and management priorities could be directed toward other ponds. For example, for Lake Wequaquet and Bearses Pond, two of Great Marsh shared ponds, PALS sampling indicated some water quality concerns (*e.g.*, occasional and regular dissolved oxygen concentrations less than MassDEP surface water regulatory minimums, respectively). Based on a request from the Town of Barnstable, CSP/SMAST staff developed a more refined May to November 2007 sampling strategy and water quality assessment that included watershed delineation, watershed land use analysis for nitrogen and phosphorus loading and development of annual water budgets, and included a comprehensive review of 2007 water quality data and all previous years water quality data (Eichner, 2009). The findings from this project led to collection of updated data in 2010 (Eichner and Howes, 2011), continuous water quality recording during the late summers of 2014 (Eichner, *et al*., 2015) and 2015 (in prep), and collection of 2015 sediment cores in Bearses Pond to assess available sediment phosphorus and the dissolved oxygen conditions that partially control the amount of release of sediment phosphorus to the water column (in prep). These more targeted data collections will allow the town to assess water quality management strategies. Collectively, this data also allowed project staff to develop pond-specific nitrogen attenuation rates in this Barnstable Great Marsh MEP assessment for these ponds: Lake Wequaquet, 86% attenuation;

Table IV-4. Freshwater Ponds within the Barnstable Great Marsh MEP Watershed.

Ponds are divided into two groups: a) ponds within the Barnstable Great Marsh MEP watershed and b) ponds on the boundary of the Great Marsh watershed and shared with other MEP watersheds. In order to alter MEP standard 50% natural nitrogen attenuation rate in freshwater ponds, a reliable bathymetric map and sufficient water quality data throughout at least one summer is generally required. Shared pond attenuated nitrogen loads to Barnstable Great Marsh subwatersheds were determined in other previous MEP assessments of Centerville River and Three Bays.

Pond	Town	PALS #	Area	Bathymetry		Max Depth	Sufficient WQ Data	WQ data review
			acres	Y/N	source	m	Y/N	
Ponds with delineated watersheds								
Bog	Barnstable	BA-382	5.6	N		-	N	No data available
Garretts	Barnstable	BA-510	27.9	Y	IEP, 1980	8.6	N	One 1980 sampling + PALS Aug/Sept
Hathaway N	Barnstable	BA-565	20.9	Y	MassDFW	17.4	N	Numerous PALS Aug/Sept
Hathaway S	Barnstable	BA-594	12.6	N		-	N	One 2001 PALS sampling
Hinckley	Barnstable	BA-411	10.3	N		6.5	N	Numerous PALS Aug/Sept
Mill	Barnstable	BA-391	16.7	N		1.4	N	3 in summer 1980; numerous PALS Aug/Sept
Dennis	Yarmouth	YA-472	47.8	Y	MassDFW	6.0	N	Numerous PALS Aug/Sept
Elishas	Yarmouth	YA-493	10.2	N		9.0	N	Numerous PALS Aug/Sept
Greenough	Yarmouth	YA-492	26.4	N		7.5	N	Numerous PALS Aug/Sept
Matthews	Yarmouth	YA-371	35.6	N		1.4	N	Numerous PALS Aug/Sept
Shared Ponds								
Wequaquet	Barnstable	BA-605	596.3	Y	IEP/KV, 1989	10.4	Y	2007 May to Nov bimonthly; WQ Assessment (Eichner, 2009) + 2010, 2014, 2015 continuous monitoring + numerous PALS Aug/Sept
Bearses	Barnstable	BA-617	66.8	Y	IEP/KV, 1989	6.2	Y	
Shallow	Barnstable	BA-626	78.4	Y	KV/IEP, 1993	2.1	N	1986 sufficient sampling + numerous PALS Aug/Sept
Lawrence	Sandwich	SA-431	133.8	Y	MassDFW	9.0	N	Numerous PALS Aug/Sept

Notes:

a) Barnstable Ponds bathymetry sources and sampling frequencies cited from Eichner (2008) with supplemental review of Cape Cod PALS results.

b) Pond areas based on Eichner, *et al.*, 2003 or MassGIS wetland coverage (1:12,000)

c) Massachusetts Division of Fisheries & Wildlife (MassDFW) maintains selected pond bathymetric maps on the division website for southeastern Massachusetts: http://www.mass.gov/eea/agencies/dfg/dfw/maps-destinations/pond-maps-southeast-district.html

d) Maximum depths are based on PALS monitoring (if available)

e) Sufficient Water Quality Data assessment is based on whether sufficient data has been gathered to adjust the standard MEP 50% natural nitrogen attenuation rate.

Bearses Pond, 84% and Shallow Pond, 84%. Shallow Pond is mostly within the Bearses Pond and the Gooseberry Pond subwatershed to Lake Wequaquet. Extensive water quality in the two Lake Wequaquet basins and older high-intensity sampling in Shallow Pond allowed CSP/SMAST staff to develop a nitrogen budget for Lake Wequaquet that included Shallow Pond (Eichner, 2009); this assessment is the basis for the Shallow Pond attenuation rate and the nitrogen load from Shallow Pond to the Great Marsh subwatersheds. Lawrence Pond in Sandwich is also a shared pond and its nitrogen load to the Great Marsh subwatersheds was previously determined in the Three Bays MEP assessment (Howes, *et al*., 2006b)

There is one additional pond where the MEP standard attenuation was adjusted, but in the case of Mill Pond, the attenuation rate was adjusted down to 26% based on stream monitoring data collected during the MEP (see Section IV.2). Mill Pond is located in the Boat Cove Creek subwatershed (#7 subwatershed), approximately 350 m upgradient/upstream of the MEP gauge. Since Mill Pond and its watershed are completely within the Creek gauge watershed, measurement of nitrogen loads at the gauge must include the attenuated nitrogen loads from the pond. Comparison of the measured and estimated loads at the gauge showed that with a standard 50% pond attenuation, the estimated watershed load was too low. Review of the available characteristics of Mill Pond showed that it is extremely shallow (maximum depth of 1.4 m). In ponds with similar depths (*e.g.*, Mill Pond in Marstons Mills (Howes, *et al.*, 2006b), Cedar Pond in North Falmouth (Howes, *et al.*, 2013), MEP measurements have shown that nitrogen attenuation rates are often lower than the standard 50%. These lower rates are likely due to shorter residence times; extensive measurements in various types of wetland systems have shown that nitrogen attenuation rates are strongly influence by residence times (*e.g.*, Johnston, 1994; Saunders and Kalff, 2001; Toet, *et al*., 2005). A reasonable estimate of the residence time of Mill Pond would be 20 days or <2% of the Lake Wequaquet residence time (Eichner, 2009). Based on the measured MEP stream data and the characteristics of the pond, the lower nitrogen attenuation rate for Mill Pond is reasonable.

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment (or scenario) of potential development and accompanying nitrogen loads within the study area watersheds. The MEP buildout is relatively straightforward and is generally completed in four steps: 1) each residential parcel classified by the town assessor as developable is identified and divided by minimum allowable lot size specified in town zoning and the resulting number of new residential units is rounded down, 2) parcels classified as developable commercial and industrial parcels by the town assessor are identified, 3) residential, commercial and industrial parcels with existing development and areas greater than twice zoning's minimum lot size are identified, divided by the minimum lot size and the resulting number of new units is rounded down, and 4) results are discussed with town staff and/or planning board members and the analysis results are modified based on local knowledge.

It should be noted that the initial MEP buildout approach is relatively simple and does not generally include any modifications/refinements for lot line setbacks, wetlands, road construction, frontage requirements, parcel shape requirements, or other more detailed zoning provisions. The MEP buildout approach also does not include potential impacts associated with the higher densities usually associated with 40B affordable housing projects. The fourth step, including the discussions with town planners, and, occasionally, town planning boards and wastewater consultants, usually leads to additional insights on developments that are planned, especially developments planned on government or public service parcels, and updates to assessor classifications, including lands purchased by the town as open space. This final step may lead to removal and/or additions to the number of parcels initially identified as developable and may include application of more detailed zoning provisions.

As an example of how the MEP approach might apply, assume an 81,000 square foot lot is classified by the town assessor as a developable residential lot (MassDOR land use code 130). This lot is divided by the 40,000 square foot minimum lot size specified in town zoning and the result is rounded down to two. As a result, two additional residential lots would be added to the subwatershed in the MEP buildout scenario. This addition could then be modified during discussion of town staff.

Other provisions of the MEP buildout assessment include town assessor classification of undevelopable lots, standard treatment of commercial and industrial properties, and assumptions for lots less than the minimum areas specified by zoning. Properties classified by the town assessors as "undevelopable" (*e.g.*, MassDOR codes 132, 392, and 442) are not assigned any development at buildout (unless revised by the town review). Commercial and industrial properties classified as developable are not subdivided; the area of each parcel and the watershed-specific factors in Table IV-2 are used to determine an estimated building size and wastewater flow for these properties. Pre-existing lots classified by the town assessor as developable are also treated as developable even if they are less than the minimum lot size specified in zoning; so, for example, a 10,000 square foot lot classified by the town assessor as a developable residential property (MassDOR 130 land use code) and located in a zoning area with a 40,000 square feet minimum lot size will be assigned an additional residential dwelling in the MEP buildout scenario. Most town zoning bylaws have a lower minimum lot size for pre-existing lots (usually 5,000 square feet) that will minimize instances of regulatory takings. Existing developed residential properties that are larger than zoning's minimum lot sizes are also assigned additional development potential only if enough area is available to accommodate at least one additional lot as specified by the zoning minimum. All of these standard approaches may be modified during consultation with town development experts prior to developing associated build-out nitrogen loadings.

Following the completion of the initial buildout assessment for the Barnstable Great Marsh watersheds, MEP staff reviewed the results with town officials. MEP staff reviewed the preliminary watershed buildout results in a series of meetings with the following town staff:

Barnstable:	Jo Anne Buntich, Director, Growth Management Department, Elizabeth Jenkins, Regulatory Planner, and Jim Benoit, GIS Director
Yarmouth:	Karen Greene, Director of Community Development and Kathy Williams, Town Planner
Dennis:	Daniel J. Fortier, Town Planner

Sandwich results were not reviewed given the small number of properties within the watershed.

Town corrections to the initial buildout estimates included removals, additions, and adjustments. All suggested changes from town staff based on the initial review were incorporated into the final MEP buildout for Barnstable Great Marsh.

All the parcels with additional buildout potential within the Barnstable Great Marsh watershed are shown in Figure IV-6. Overall, this buildout includes a projected 1,639 additional residences, 1,821,606 square feet of additional commercial properties and 3,370,249 square feet of additional industrial properties. In the buildout nitrogen loading scenario, each additional residential, commercial, or industrial property added at buildout is assigned nitrogen loads for

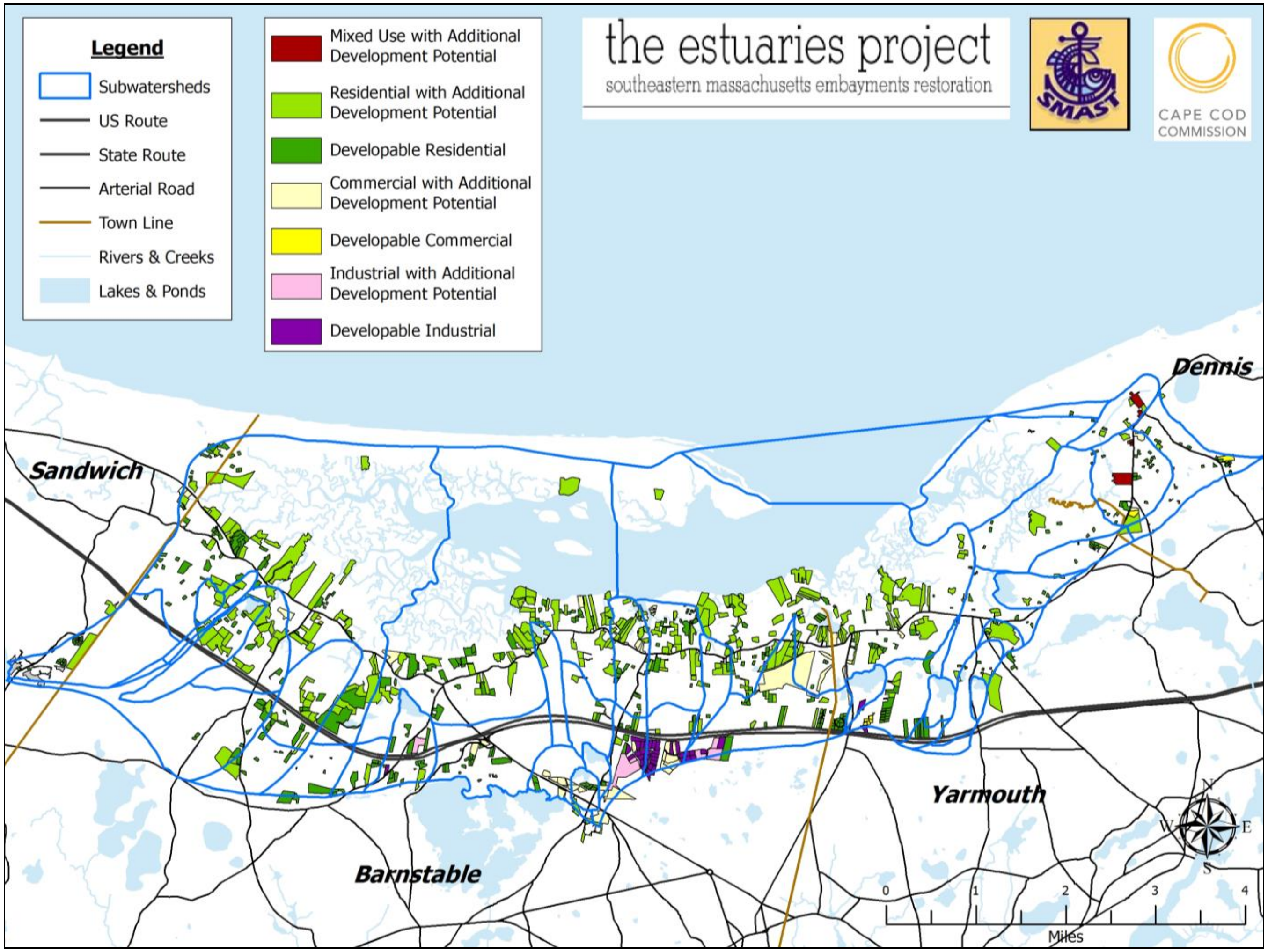


Figure IV-6. Developable Parcels in the Barnstable Great Marsh watershed. Indicated parcels are assigned additional nitrogen loads in the MEP buildout scenario. Buildout is based on existing town zoning and town assessor classifications. Parcels colored light green, red, light yellow, and pink are existing developed parcels (residential, mixed use, commercial and industrial, respectively) with additional development potential based on current zoning, while parcel colored dark green, yellow, and purple are undeveloped parcels classified as developable by the respective town assessors (residential, commercial and industrial, respectively). These buildout results include adjustments indicated by town officials during review of initial MEP buildout estimates.

wastewater and impervious surfaces. Residential additions also include lawn fertilizer nitrogen additions. All wastewater loads are assumed to come from standard on-site septic systems unless the parcel is designated as already having a sewer connection (for additional development on existing lots) or identified within the sewer service area; all properties with the sewer service area are assumed to connect to the Hyannis sewer system and have no assigned wastewater nitrogen loads within the watershed. Cumulative unattenuated buildout loads are indicated in a separate column in Table IV-3. It should be noted that this is one example of a buildout scenario; alternative assumptions about future development could be developed to assess the water quality impacts of other buildout scenarios. Based on the MEP assessment, buildout additions within the Barnstable Great Marsh watersheds will increase the unattenuated watershed nitrogen loading rate by 22%.

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

IV.2.1 Background and Purpose

Modeling and predicting changes in coastal embayment nitrogen related water quality is based, in part, on determination of the inputs of nitrogen from the surrounding contributing land or watershed relative to the tidal flushing and nitrogen cycling within the embayment basins. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out, sewering analysis, enhanced flushing, pond/wetland restoration for natural attenuation, etc.) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of the Barnstable Great Marsh estuary system being investigated under this nutrient threshold analysis were based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1).

If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the estuarine receiving waters. This condition exists in watersheds where nitrogen transport from source to estuarine waters is through groundwater flow in sandy outwash aquifers (such being the case in the developed region of southeastern Massachusetts but more so on Cape Cod). The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) can be diminished by natural biological processes that represent removal (not just temporary storage). However, this potential natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes to varying degrees based on habitat and residence time. In the watershed for the Barnstable Great Marsh Estuary, a portion of the freshwater flow and transported nitrogen passes through several surface water systems (e.g. Alder Brook, Boat Cove Creek, Bridge Creek, Maraspin Creek, Whites Brook and Chase Garden Creek) prior to entering the estuary, producing the opportunity for significant nitrogen attenuation under appropriate conditions (Figure IV-7). It should be noted that during the MEP stream gauging assessment, two additional locations were evaluated (heads of Scorton Creek and Brickyard Creek), however, flow in Scorton Creek was too low to measure accurately and Brickyard Creek was dry through 95% of the assessment period. Those locations are depicted in Figure IV-7 but were not included in the MEP assessment.

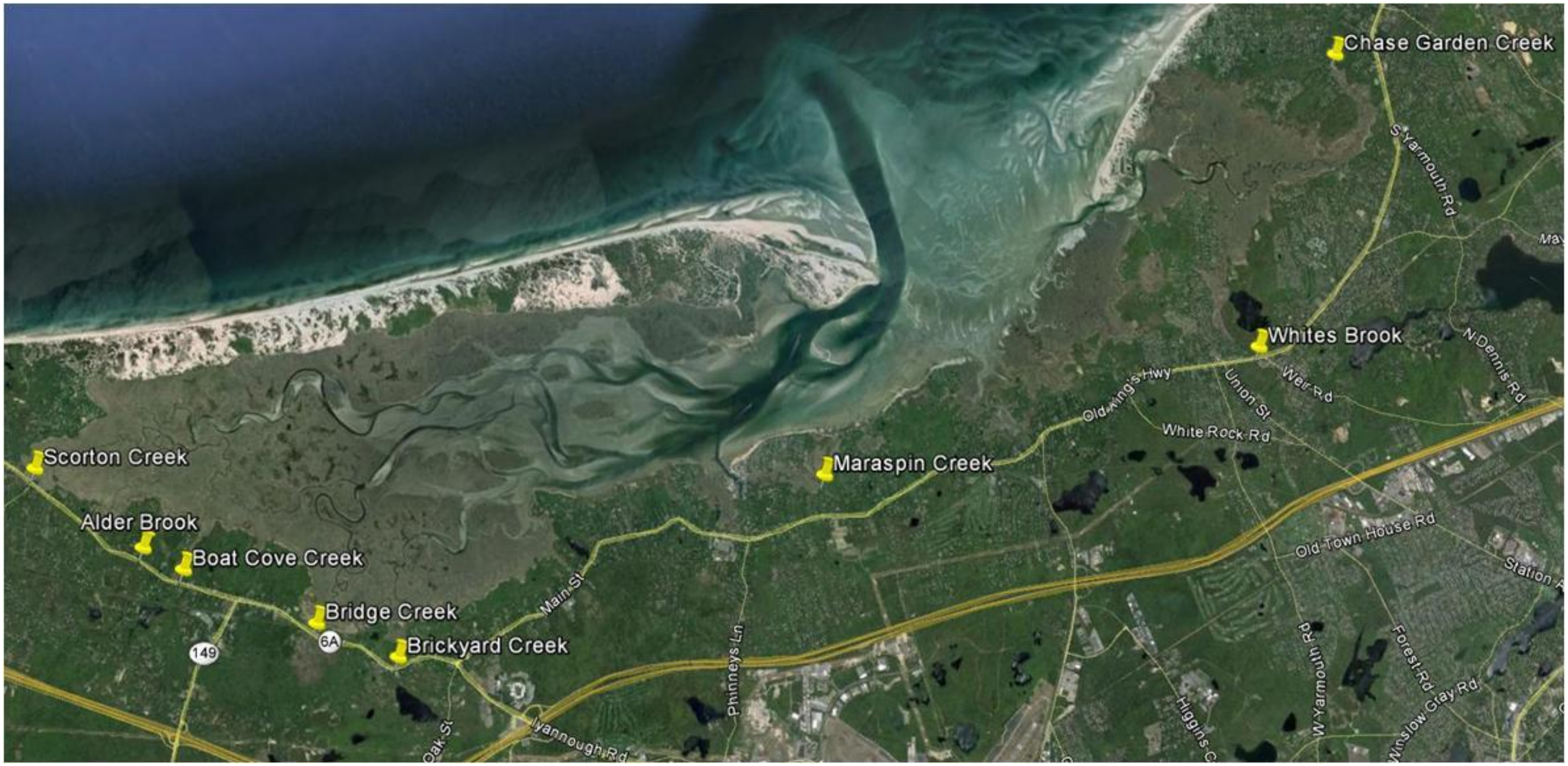


Figure IV-7. Location of Stream gauges (yellow symbols) in the Barnstable Great Marsh estuary system. Two stream gauge locations (Scorton Creek and Brickyard Creek) did not have measureable flow during the gauge deployment period and therefore could not be used for direct measurement of surfacewater load. Rather, loads from those areas of the watershed were considered as being transported to Barnstable Harbor via groundwater.

Failure to determine the attenuation of watershed derived nitrogen overestimates the nitrogen load to receiving estuarine waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). In addition to attenuation by freshwater ponds (see Section IV.1.3, above), attenuation in surface water flows is also important. An example of the significance of surface water nitrogen attenuation relating to embayment nitrogen management was seen in the Agawam River, where >50% of nitrogen originating within the upper watershed was attenuated in ponds and streams prior to discharge to the Wareham River Estuary (CDM 2000). Similarly, MEP analysis of the Quashnet River (Town of Falmouth, Cape Cod) indicated that in the upland watershed, which has natural attenuation predominantly associated with riverine processes, the integrated attenuation was 39% (Howes et al. 2004). In addition, a preliminary study of Great, Green and Bournes Ponds in Falmouth, measurements indicated a 30% attenuation of nitrogen during stream transport (Howes and Ramsey 2001). An example where natural attenuation played a significant role in nitrogen management can be seen relative to West Falmouth Harbor (Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater effluent plume emanating from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Therefore, proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach in the Barnstable Great Marshes overall watershed. MEP conducted long-term measurements of natural attenuation relating to the most significant surface water discharges to the estuary in addition to the natural attenuation measures by fresh kettle ponds, addressed above (Section IV.1). These additional site-specific studies were conducted in the 6 major surface water flow systems in the Barnstable Great Marsh estuary system watershed, 1) Alder Brook discharging to the Great Marshes, 2) Boat Cove Brook discharging to the Great Marshes, 3) Bridge Creek discharging to the Great Marshes, 4) Maraspin Creek discharging to the Barnstable Harbor (Millway), 5) Whites Brook discharging to Bass Hole and 6) Chase Garden Creek discharging to Bass Hole. A stream gauge was deployed at the head of Scorton Creek, however flows were too low to measure accurately therefore it was discontinued as a stream gauging site. Similarly, regarding Brickyard Creek, given the small size of the drainage area as well as the intermittent flow (the creek channel was dry 95% of the stream gauge deployment period) by comparison to the other six major surface water discharges measured by the MEP, the technical team agreed to exclude Brickyard Creek from the gauging program.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the freshwater streams discharging to the estuary provides a direct integrated measure of all of the processes presently attenuating nitrogen in the contributing area up-gradient from the various gauging sites. Flow and nitrogen load were measured at the gauges in each freshwater stream site for between 16 and 24 months depending on the stream gauging location (Figures IV-8 to IV-18). During each study period, volumetric discharge measurements were completed on each surface water inflow every month to two months. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream volumetric flow (Q).

Determination of stream flow at each gauge was calculated and based on the measured values obtained for stream cross sectional area and velocity. Stream discharge was represented by the summation of individual discharge calculations for each stream subsection for which a cross sectional area and velocity measurement were obtained. Velocity measurements across the entire stream cross section were not averaged and then applied to the total stream cross sectional area.

The formula that was used for calculation of stream flow (discharge) is as follows:

$$Q = \Sigma(A * V)$$

where by:

Q = Stream discharge (m^3/s)
A = Stream subsection cross sectional area (m^2)
V = Stream subsection velocity (m/s)

Thus, each stream subsection will have a calculated stream discharge value and the summation of all the sub-sectional stream discharge values will be the total calculated discharge for the stream.

Periodic measurement of flows over the entire stream gauge deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gauges. Water level data obtained every 10-minutes was averaged to obtain hourly stages for a given river/stream/creek/brook. These hourly stages values where then entered into the stage-discharge relation to compute hourly flow. Hourly flows were summed over a period of 24 hours to obtain daily flow and further, daily flows summed to obtain annual flow. In the case of tidal influence on stream stage, the diurnal low tide stage value was extracted on a day-by-day basis in order to resolve the stage value indicative of strictly freshwater flow. The lowest low tide stage values for any given day were utilized in the stage – discharge relation in order to compute daily flow as this stage value is most representative of freshwater flow. A complete annual record of stream flow (365 days) was generated for the surface water discharges flowing into the Barnstable Great Marsh Estuarine System.

The annual flow record for the surface water flow at each gauge was merged with the nutrient data from the weekly water quality sampling performed at each gauge location to determine surfacewater related nitrogen loading rates to the Barnstable Great Marshes Estuary. The nitrogen load discharged from the streams was calculated using the paired daily volumetric discharge and daily nitrogen concentration measurements to determine the mass flux of nitrogen through each specific gauge site. For each of the stream gauge locations, weekly water samples were collected (at low tide for a tidally influenced stage) in order to determine nutrient concentrations from which nutrient load was calculated. In order to pair daily flows with daily nutrient concentrations, interpolation between weekly nutrient data points was necessary. These data are expressed as nitrogen mass per unit time (kg/d) and can be summed in order to obtain weekly, monthly, or annual nutrient loads to the embayment system as appropriate. Comparing these measured nitrogen loads based on stream flow and water quality sampling to predicted loads based on the land use analysis allowed for the determination of the degree to which natural biological processes within the watershed to each gauged stream currently reduces (percent attenuation) nitrogen loading to the overall embayment systems.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Alder Brook Discharge to Barnstable Harbor

Similar to other surface water features in the MEP study region that typically emanate from a specific pond or wetland, Alder Brook, which discharges into the Great Marshes, has a clearly demarcated up-gradient bog/wetland area from which the small brook discharges. Based on numerous previous studies completed by the MEP on other systems in southeastern Massachusetts, the outflow from the bog/wetlands and the wooded areas up-gradient of the Alder Brook gauge very likely contribute to the attenuation of nitrogen during transport, while the stream provides a watershed “drain” that allows for direct measurement of the net nitrogen load, hence nitrogen attenuation. The combined rate of nitrogen attenuation by the biological processes that occur in the various surface water features to each stream site was determined by comparing the present predicted nitrogen loading from land use analysis within the upland contributing area to the bog/wetlands and wooded areas above the gauge site and the measured annual discharge of nitrogen to the downgradient Great Marshes, Figure IV-8.

At the Alder Brook gauge site (established at culvert passing under Route 6A), a continuously recording vented calibrated water level gauge was installed to yield the level of water in the channel that carries the flows and associated nitrogen load to the estuarine system. As the lower reach of Alder Brook is tidally influenced down gradient of Route 6A, the stage record from the gauge was checked to make sure there was no tidal influence in the record at low tide. To confirm that freshwater was being measured, the stage record was analyzed for any semi-diurnal variations indicative of tidal influence and salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average salinity of the water samples taken from Alder Brook at Route 6A at low tide confirmed that the stream was carrying fresh water at the gauge site ($\leq$0.1 ppt). Therefore, the gauge location was deemed acceptable for making freshwater flow measurements at low tide. Calibration of the gauge was checked monthly. The gauge on Alder Brook was installed on June 12, 2006 and was set to operate continuously for a complete hydrologic year (low flow to low flow, ~12 months). Stage data collection continued until September 18, 2007 for a total deployment of 15 months.

Surface freshwater flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for Alder Brook at the Route 6A gauge site based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the estuary and is reflective of the biological processes occurring in the stream channel and the network of bogs/wetlands and wooded area contributing to nitrogen attenuation (Figure IV-9 and Table IV-5a, b and IV-6a, b). In addition, a water balance was constructed based upon the U.S. Geological Survey/MEP/CCC defined watershed delineations to determine long-term average freshwater discharge expected at each gauge site based on area and average recharge.

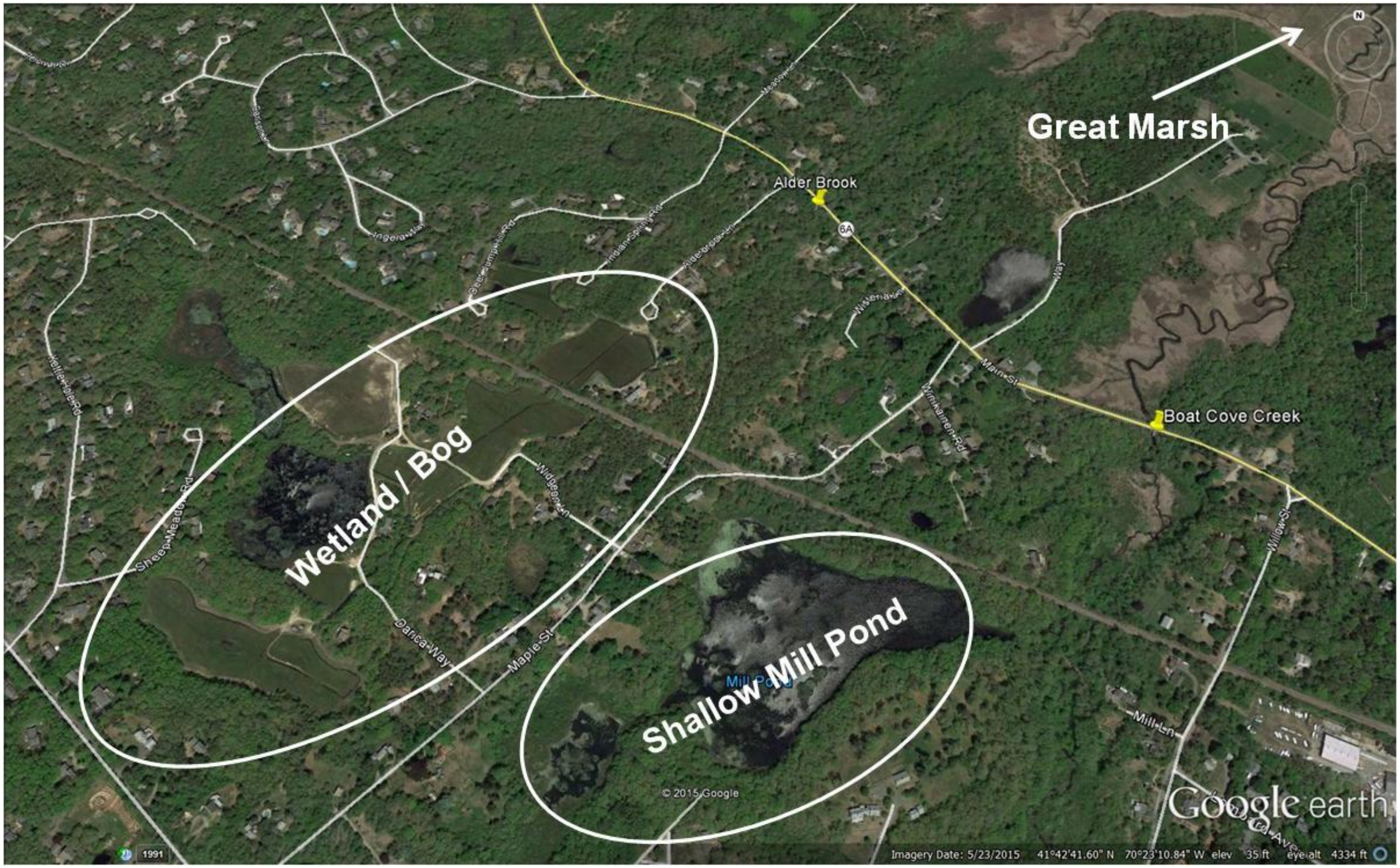


Figure IV-8. Location of MEP stream gauges (yellow symbol) for measuring flow and nitrogen loads transported by Alder Brook and Boat Cove Creek to estuarine waters. Alder Brook receives surfacewater from a upgradient bog/wetland feature whereas Boat Cove Creek discharges from a shallow up-gradient pond.

The annual freshwater flow record for Alder Brook as measured by the MEP was compared to the long-term average flows determined by the USGS/MEP/CCC modeling effort (Table III-1). The measured freshwater discharge from Alder Brook at the Route 6A gauge location was only ~3% below the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 917 m^3/day compared to the long term average flow of 943 m^3/day determined by the watershed modeling effort. The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Alder Brook discharging from the sub-watershed indicates that the Brook is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Alder Brook outflow were moderate, 0.857 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 0.79 kg/day and a measured total annual TN load of 287 kg/yr. In the Alder Brook flow, nitrate made up a very small fraction (9%) of the total nitrogen pool, indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the bog/wetland areas and stream bed up-gradient of the gauge was being transformed by plants and microbes within the surface water bog/wetlands upgradient of the gauge site. Given the relatively low levels of remaining nitrate in the stream discharge, the possibility for additional uptake by freshwater systems appears to be limited in the Alder Brook sub-watershed.

From the measured nitrogen load discharged by Alder Brook to the Great Marshes and the nitrogen load determined from the stream watershed based land use analysis, it appears that there is moderate nitrogen attenuation of upper watershed derived nitrogen during transport to the down gradient estuary. Based upon the slightly lower total nitrogen load (287 kg yr^{-1}) discharged from Alder Brook at Route 6A compared to that added by the various land-uses to the associated watershed (485 kg yr^{-1}), there was a 41% integrated attenuation of nitrogen in passage through the stream and up-gradient freshwater wetlands prior to discharge to the estuary (i.e. 41% of nitrogen input to watershed does not reach the estuary). This level of attenuation is consistent with other streams evaluated under the MEP with up-gradient wetlands/bogs capable of attenuating nitrogen. The directly measured nitrogen load from Alder Brook was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

Table IV-5a. Comparison of water flow and nitrogen load discharged by surface waters (freshwater) to the Barnstable Great Marshes. The "Stream" data are from the MEP stream gauging effort. Watershed data are based upon the MEP watershed land use modeling effort (Section IV.1) and the combination of USGS watershed delineations and watershed delineation information provided by the CCC. Delineations were reviewed by MEP Technical Team Members and sub-watershed delineations were developed by the MEP (Section III).

Stream Discharge Parameter	Alder Brook Discharge[a] Great Marsh	Boat Cove Creek Discharge[a] Great Marsh	Bridge Creek Discharge[a] Great marsh	Maraspin Creek Discharge[a] Barnstable Hrb.	Data Source
Total Days of Record	365[b]	365[b]	365[b]	365[b]	(1)
Flow Characteristics					
Stream Average Discharge (m3/day)	917	12,850	5,472	1,384	(1)
Contributing Area Average Discharge (m3/day)	943	12435	5668	1396	(2)
Discharge Stream 2006-07 vs. Long-term Discharge	-2.84%	3.23%	-3.58%	-0.87%	
Nitrogen Characteristics					
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.078	0.261	0.171	1.659	(1)
Stream Average Total N Concentration (mg N/L)	0.857	0.685	0.648	2.114	(1)
Nitrate + Nitrite as Percent of Total N (%)	9%	38%	26%	78%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	0.79	8.797	3.55	2.93	(1)
TN Average Contributing UN-attenuated Load (kg/day)	1.33	8.53	5.97	3.73	(3)
Attenuation of Nitrogen in Pond/Stream (%)	41%	-3%	41%	21%	(4)

(a) Flow and N load to streams discharging to Barnstable Harbor, the Great Marsh and Bass Hole include apportionments of Pond contributing areas as appropriate.
(b) Average September 1, 2006 to August 31, 2007.

(1) MEP gage site data
(2) Calculated from MEP watershed delineations to ponds upgradient of specific gages;
the fractional flow path from each sub-watershed which contribute to the flow in the streams to Barnstable Harbor-Bass Hole;
and the annual recharge rate.
(3) As in footnote (2), with the addition of pond and stream conservative attentuation rates.
(4) Calculated based upon the measured TN discharge from the rivers vs. the unattenuated watershed load.

Table IV-5b. Comparison of water flow and nitrogen load discharged by surface waters (freshwater) to the Bass Hole sub-estuary. The "Stream" data are from the MEP stream gauging effort. Watershed data are based upon the MEP watershed land use modeling effort (Section IV.1) and the combination of USGS watershed delineations and watershed delineation information provided by the CCC. Delineations were reviewed by MEP Technical Team Members and sub-watershed delineations were developed by the MEP (Section III).

Stream Discharge Parameter	**Whites Brook Discharge[(a)]** Bass Hole	**Chase Garden Creek Discharge[(a)]** Bass Hole	**Data Source**
Total Days of Record	365[(b)]	365[(b)]	(1)
Flow Characteristics			
Stream Average Discharge (m3/day)	1,462	1,611	(1)
Contributing Area Average Discharge (m3/day)	1,379	1,574	(2)
Discharge Stream 2006-07 vs. Long-term Discharge	5.68%	2.30%	
Nitrogen Characteristics			
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.542	0.371	(1)
Stream Average Total N Concentration (mg N/L)	0.959	1.058	(1)
Nitrate + Nitrite as Percent of Total N (%)	57%	35%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	1.403	1.704	(1)
TN Average Contributing UN-attenuated Load (kg/day)	2.24	4.28	(3)
Attenuation of Nitrogen in Pond/Stream (%)	37%	60%	(4)

(a) Flow and N load to streams discharging to Bass Hole include apportionments of Pond contributing areas as appropriate.
(b) Average September 1, 2006 to August 31, 2007.

(1) MEP gage site data
(2) Calculated from MEP watershed delineations to ponds upgradient of specific gages;
the fractional flow path from each sub-watershed which contribute to the flow in the streams to Barbstable Harbor-Bass Hole;
and the annual recharge rate.
(3) As in footnote (2), with the addition of pond and stream conservative attentuation rates.
(4) Calculated based upon the measured TN discharge from the rivers vs. the unattenuated watershed load.

Table IV-6a. Summary of annual volumetric discharge and nitrogen load from the four major surface water discharges to the Barnstable Great Marsh estuary system (based upon the data presented in Figures IV-9, 10, 12,14 and Table IV-5a.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Barnstable Harbor - Great Marsh Alder Brook MEP measured	**September 1, 2006 to August 31, 2007**	**334,705**	**26**	**287**
Barnstable Harbor - Great Marsh Alder Brook	Based on Watershed Area and Recharge	**344,195**	--	--
Barnstable Harbor - Great Marsh Boat Cove Creek MEP measured	**September 1, 2006 to August 31, 2007**	**4,690,250**	**1,223**	**3,211**
Barnstable Harbor - Great Marsh Boat Cove Creek	Based on Watershed Area and Recharge	**4,538,775**	--	--
Barnstable Harbor - Great Marsh Bridge Creek MEP measured	**September 1, 2006 to August 31, 2007**	**1,997,280**	**342**	**1,294**
Barnstable Harbor - Great Marsh Bridge Creek	Based on Watershed Area and Recharge	**2,068,820**	--	--
Barnstable Harbor - Millway Maraspin Creek MEP measured	**September 1, 2006 to August 31, 2007**	**505,160**	**838**	**1,051**
Barnstable Harbor - Millway Maraspin Creek	Based on Watershed Area and Recharge	**509,540**	--	--

Table IV-6b. Summary of annual volumetric discharge and nitrogen load from the four major surface water discharges to the Barnstable Great Marsh estuary system (based upon the data presented in Figures IV-16,18 and Table IV-5b.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Barnstable Harbor - Bass Hole Chase Garden Creek MEP measured	**September 1, 2003 to August 31, 2004**	**588,172**	**218**	**622**
Barnstable - Bass Hole Chase Garden Creek	Based on Watershed Area and Recharge	**576,600**	--	--
Barnstable Harbor - Bass Hole Whites Brook MEP measured	**September 1, 2003 to August 31, 2004**	**533,627**	**289**	**512**
Barnstable Harbor - Bass Hole Whites Brook	Based on Watershed Area and Recharge	**503,335**	--	--

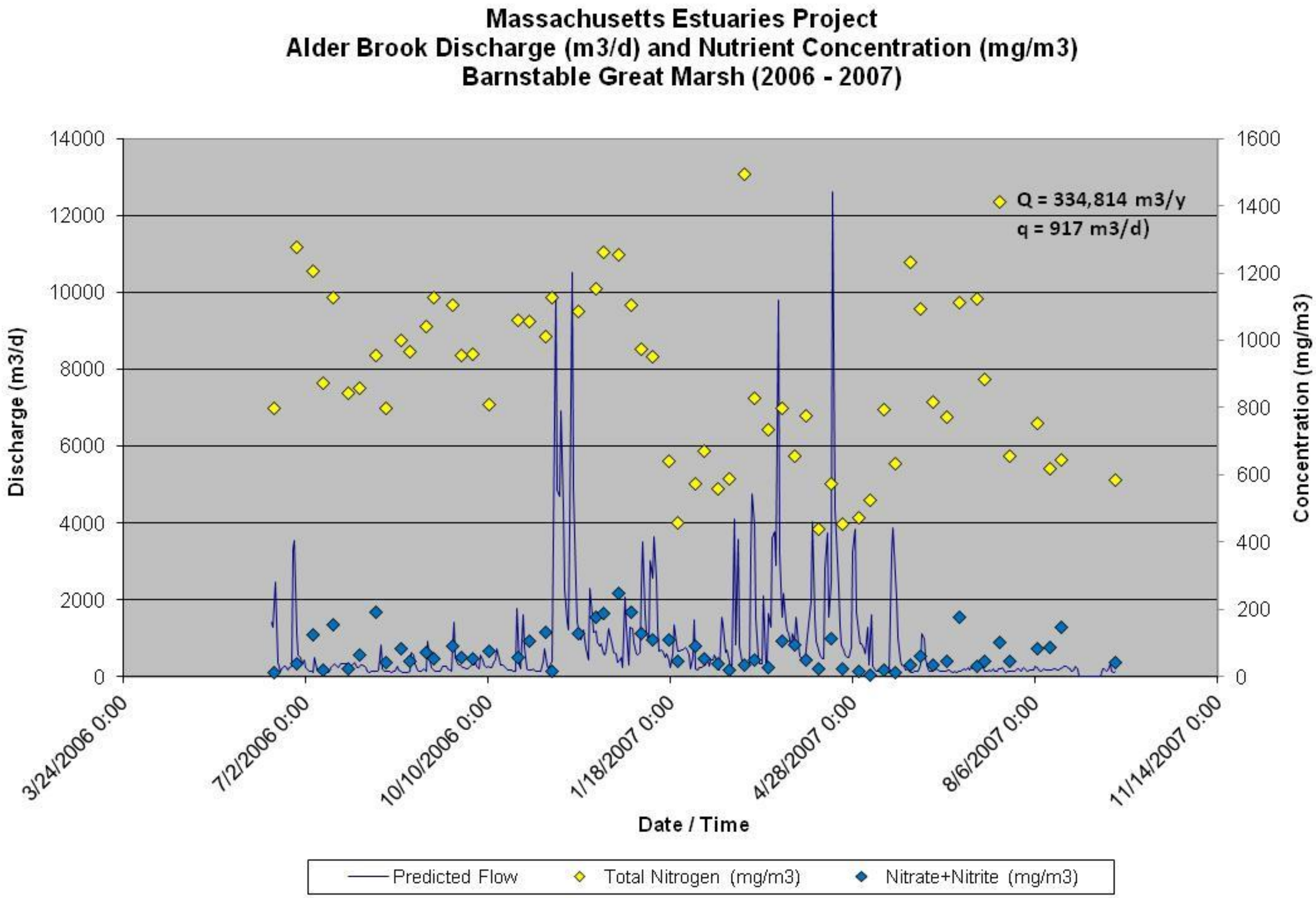


Figure IV-9. Discharge from Alder Brook to Barnstable Great Marshes (solid blue line). Total nitrogen (yellow symbols) and Nitrate + Nitrite (NOx, blue symbols) concentrations are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-3).

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Boat Cove Creek Discharge to Barnstable Harbor

Similar to other surface water features in the MEP study region that typically emanate from a specific pond or wetland, Boat Cove Brook, which discharges into the Great Marshes, does have a clearly demarcated up-gradient mill pond from which the small brook originates. Based on numerous previous studies completed by the MEP on other systems in southeastern Massachusetts, the shallow mill pond up-gradient of the Boat Cove Brook gauge likely contributes to the attenuation of nitrogen discharged from the watershed and outflowing to the Creek. The combined rate of nitrogen attenuation by the biological processes that occur in the various surface water features of the subwatershed was determined by comparing the present predicted load from the land use analysis of the sub-watershed region contributing to the mill pond and areas above the gauge site and the measured annual discharge of nitrogen to the Great Marshes, Figure IV-8.

The freshwater flow carried by Boat Cove Brook to the estuarine waters of the Great Marshes was determined using a continuously recording vented calibrated water level gauge deployed where the brook passes through a culvert under Route 6A. As this surface water system was potentially tidally influenced, the creek discharge was checked to confirm the extent of tidal influence and whether freshwater flow could be measured at low tide in the estuary. To confirm that freshwater was being measured, salinity measurements were conducted on weekly water quality samples collected from the gauge site. Average measured sample salinity confirmed that the stream was carrying fresh water at the gauge site ($\leq$0.1 ppt) and was clearly not tidally influenced. As such, a salinity adjustment was not necessary in order to determine daily flows using the MEP developed stage-discharge relation. The Boat Cove Brook gauge location was deemed acceptable for measurements of annual freshwater flow. Calibration of the gauge was checked monthly. The gauge was installed on June 11, 2006 and was set to operate continuously for 16 months such that at least one summer season would be captured in the flow record. Stage data collection continued until December 13, 2007 for a total deployment of 18 months.

Stream flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gauge site based upon these flow measurements and the measured water levels at the gauge site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the Barnstable Great Marsh estuary system and reflective of the biological processes occurring in the stream channel, wetlands and wooded areas contributing to nitrogen attenuation (Figure IV-10 and Tables IV-5a, b and IV-6a, b). In addition, a water balance was constructed based upon the U.S. Geological Survey/CCC/MEP defined watershed delineations to determine long-term average freshwater discharge expected at the Boat Cove Brook gauge site based on area and average recharge.

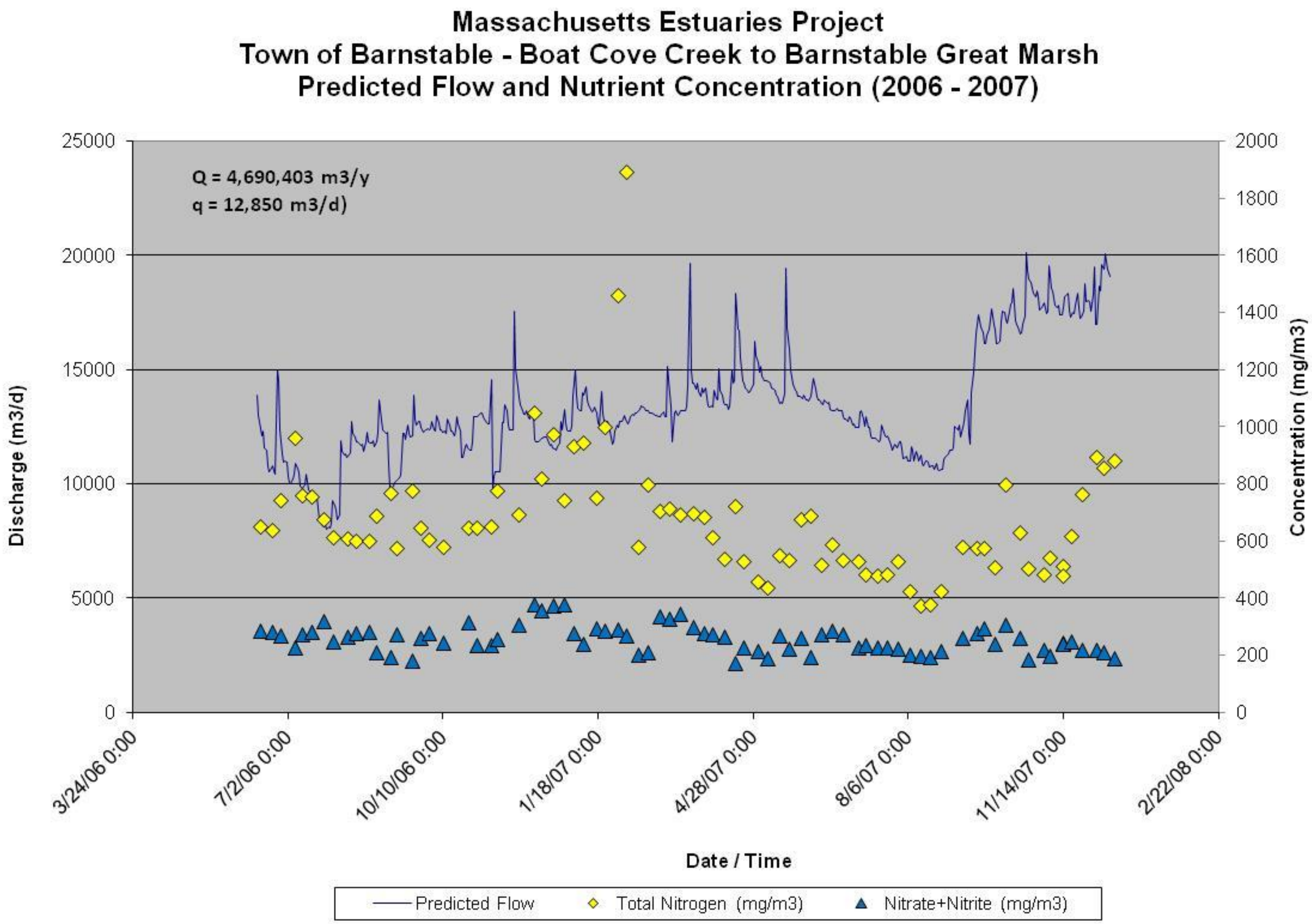


Figure IV-10. Discharge from Boat Cove Brook to the Great Marshes (solid blue line). Total nitrogen (yellow symbols) and Nitrate + Nitrite (NOx, blue symbols) concentrations are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-3).

The annual freshwater flow record for the Boat Cove Brook as measured by the MEP was compared to the long-term average flows determined by the USGS/MEP/CCC modeling effort (Table III-1). The measured freshwater discharge from the Boat Cove Brook at the Route 6A gauge location was only ~3% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 12,850 m^3/day compared to the long term average flows determined by the watershed modeling effort of 12,435 m^3/day. The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Boat Cove Brook discharging from the sub-watershed indicates that the Brook is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Boat Cove Brook outflow were low to moderate, 0.685 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 8.8 kg/day and a measured total annual TN load of 3,211 kg/yr. In Boat Cove Brook, nitrate made less than half of the total nitrogen pool (38%), indicating that a large portion of the groundwater nitrogen (typically dominated by nitrate) discharging to the wetland areas and stream bed up-gradient of the gauge was being taken up by plants and transformed within these different aquatic systems. Given the moderate levels of remaining nitrate in the stream discharge, the possibility for additional uptake by freshwater systems may be possible in the Boat Cove Brook sub-watershed should the Town of Barnstable (through management) want to restore the shallow mill pond that is the primary source water for the brook.

From the measured nitrogen load discharged by Boat Cove Brook to the Great Marshes and the nitrogen load determined from the watershed based land use analysis, it appears that there is insignificant nitrogen attenuation of upper watershed derived nitrogen during transport through the mill pond and Boat Cove Brook discharging to the estuary. Based upon the similar total nitrogen load (3,211 kg yr^{-1}) discharged from Boat Cove Brook at Route 6A compared to that added by the various land-uses within the contributing areas to the associated watershed (3,114 kg yr^{-1}), the integrated attenuation in passage through the stream and up-gradient freshwater wetlands prior to discharge to the estuary is essentially zero (i.e. nitrogen input to watershed reaches the estuary with no attenuation). This lack of attenuation compared to other streams evaluated under the MEP is consistent with the nature of the shallow up-gradient mill pond and wooded areas which lack significant natural mechanisms capable of attenuating nitrogen. Surfacewater and nitrogen load traveling through this small sub-watershed has very little time during transport to interact with biological processes that could attenuate the TN-load. The directly measured nitrogen load from Boat Cove Brook was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

IV.2.4 Surface water Discharge and Attenuation of Watershed Nitrogen: Bridge Creek Discharge to Barnstable Harbor

Unlike most surface water features in the MEP study region that typically emanate from a specific pond, Bridge Creek, which discharges into the Barnstable Great Marshes, does not have an up-gradient source water pond. Rather, this small creek appears to be groundwater fed and emanates from a wetland/salt marsh area up-gradient of Route 6A. Bridge Creek up-gradient of the gauge located at Route 6A is the terminal end of a tidal creek that flows out of a tributary marsh to the larger Barnstable Great Marshes. The outflow leaving the salt marsh area up-gradient of Route 6A serves as a mechanism for directly measuring potential nitrogen attenuation resulting from natural processes taking place in the salt marsh upgradient and bordering freshwater wetland areas. The combined rate of nitrogen attenuation by the biological processes occurring as the water in Bridge Creek flows to the estuary was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to salt marsh areas and the creek channel above the gauge site and the measured annual discharge of nitrogen to the estuary at the Bridge Creek gauge, Figure IV-11.

The freshwater flow carried by Bridge Creek to the estuarine waters of the Barnstable Great Marshes was determined using a continuously recording vented calibrated water level gauge. As this surface water system was tidally influenced, the creek discharge was checked to confirm the extent of tidal influence and whether freshwater flow could be measured at low tide in the estuary. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gauge site at low tide. Average measured sample salinity was found to be 1.2 ppt, sufficiently fresh to be representative of water and load generated from the up-gradient subwatershed. As such, a salinity adjustment was not necessary in order to determine daily flows using the MEP developed stage-discharge relationship. The Bridge Creek gauge location was deemed acceptable for making flow measurements at low tide and obtaining an estimate of annual freshwater flow from the up-gradient subwatershed. Calibration of the gauge was checked monthly. The gauge was installed on June 17, 2006 and was set to operate continuously for a complete hydrologic year (low flow to low flow, ~12 months). Stage data collection continued until September 12, 2007 for a total deployment of 15 months.

Flow in the creek (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gauge site based upon these flow measurements and the measured water levels at the gauge site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the estuarine system and that is reflective of the biological processes occurring in the stream channel, the surrounding salt marsh and the wooded natural areas contributing to nitrogen attenuation (Figure IV-12 and Tables IV-5a, b and IV-6a, b). In addition, a water balance was constructed based upon the U.S. Geological Survey/MEP/CCC defined watershed delineations to determine long-term average freshwater discharge expected at the Bridge Creek gauge site based on area and average recharge.

The annual freshwater flow record for Bridge Creek as measured by the MEP was compared to the long-term average flows determined by the USGS/MEP/CCC modeling effort (Table III-1). The measured freshwater discharge from Bridge Creek at the gauge location (Route 6A) was only ~4% below the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 5,472 m^3/day compared to the long term average flows of 5,668 m^3/day determined from the watershed modeling effort. The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Bridge Creek discharging from the sub-watershed indicates that the creek is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Bridge Creek outflow were low to moderate, 0.648 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 3.55 kg/day and a measured total annual TN load of 1,294 kg/yr. In the Bridge Creek outflow, nitrate made up less than half of the total nitrogen pool (26%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the wetland areas and stream bed up-gradient of the gauge is being taken up by plants within the salt marsh and the creek bed and being converted to organic forms. Given the low levels of remaining nitrate in the creek discharge, the possibility for additional uptake by

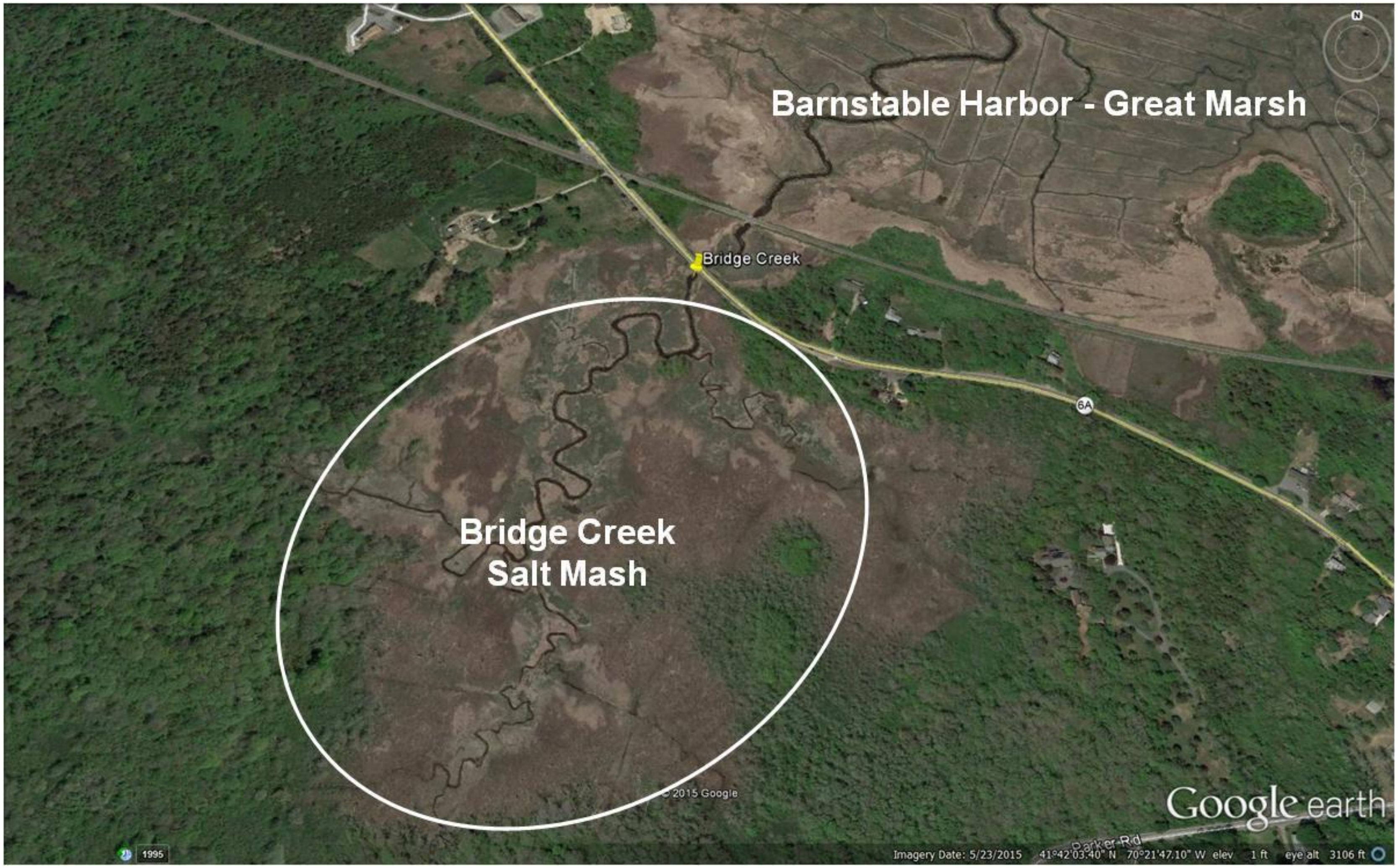


Figure IV-11. Location of MEP stream gauge (yellow symbol) for measuring flow and nitrogen load in Bridge Creek. The Bridge Creek salt marsh receives groundwater flow from surrounding wooded upland and discharges to the down gradient estuarine receiving waters of Barnstable Harbor-Great Marsh.

freshwater systems is limited, even if the Bridge Creek sub-watershed was appropriately structured or had potential aquatic features which could be enhanced or restored to facilitate denitrification.

From the measured nitrogen load discharged by Bridge Creek to the Great Marshes and the nitrogen load determined from the watershed based land use analysis, it appears that there is modest nitrogen attenuation of upper watershed derived nitrogen during transport through Bridge Creek to the estuary. Based upon the lower total nitrogen load (1,294 kg yr^{-1}) discharged from Bridge Creek compared to that added by the various land-uses to the associated watershed (2,180 kg yr^{-1}), a 41% integrated attenuation was calculated in passage through this small tidal creek and the up-gradient salt marsh prior to discharge to the estuary (i.e. 41% of nitrogen input to watershed does not reach the estuary). This level of attenuation is consistent with other streams/creeks evaluated under the MEP with similar aquatic resources (wetland/salt marsh areas). The directly measured nitrogen load from Bridge Creek was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

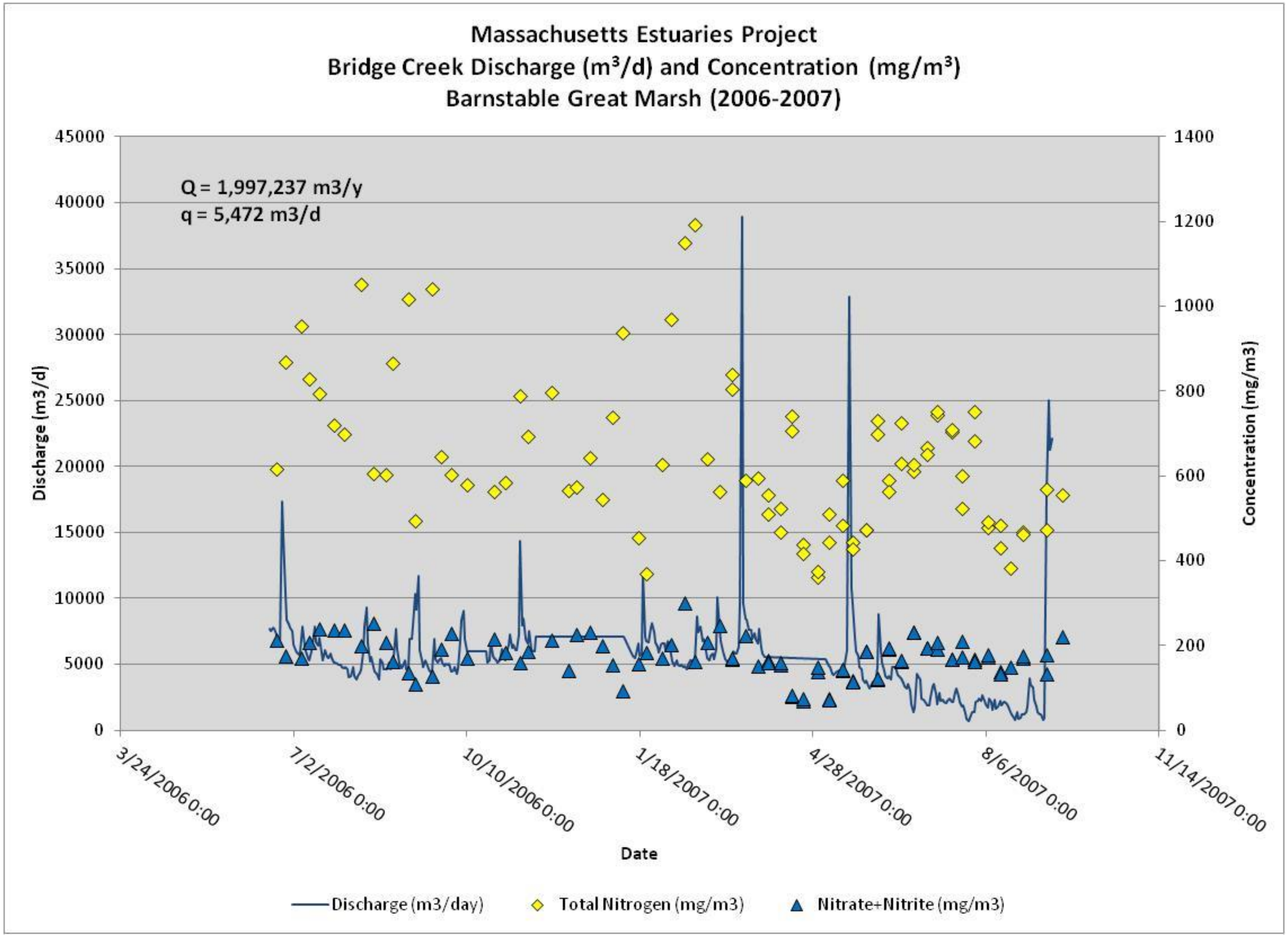


Figure IV-12. Discharge from Bridge Creek to the Barnstable Great Marshes (solid blue line). Total nitrogen (yellow symbols) and Nitrate + Nitrite (NOx, blue symbols) concentrations are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-3)

IV.2.5 Surface water Discharge and Attenuation of Watershed Nitrogen: Maraspin Creek Discharge to Barnstable Harbor-Millway

Unlike most surface water features in the MEP study region that typically emanate from a specific pond, Maraspin Creek, which discharges into Barnstable Harbor, does not have an up-gradient source water pond. Rather, this creek appears to be groundwater fed and emanates from a wetland/salt marsh area bounded by Mill Way and Commercial Road (Figure IV-6c). As the main stem of Maraspin Creek is in a salt marsh with a central tidal channel, the MEP focused its assessment of this surfacewater feature on the upper most reach of the creek up-gradient of Commercial Road. Maraspin Creek up-gradient of the gauge located at Commercial Road is the terminal end of a tidal creek that flows out of a tributary salt marsh into the Millway portion of Barnstable Harbor and is the most likely area to make flow measurements that would be representative of freshwater at low tide. The outflow leaving the small wetland area up-gradient of Commercial Road serves as a mechanism for directly measuring potential nitrogen attenuation resulting from natural processes taking place in the wetland feeding the down gradient salt marsh. The combined rate of nitrogen attenuation by the biological processes occurring as the water in Maraspin Creek flows to the estuarine waters of Barnstable Harbor was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the wetland and creek channel above the gauge site and the measured annual discharge of nitrogen to the estuary at the Maraspin Creek gauge site at Commercial Road (Figure IV-13).

The freshwater flow carried by the uppermost reach of Maraspin Creek and discharging to the estuarine waters of Barnstable Harbor was determined using a continuously recording vented calibrated water level gauge. As this surface water system was potentially tidally influenced at the Commercial Road crossing where the stream gauge was located, the creek discharge was checked to confirm the extent of tidal influence and whether freshwater flow could be measured at low tide in the estuary. To confirm that freshwater was being measured, salinity measurements were conducted on weekly water quality samples collected from the gauge site. Average measured sample salinity at low tide was found to be only 0.3 ppt, indicating a insignificant tidal influence at the gauge location at low tide. As such, salinity adjustment was not necessary to the flows in order to determine daily freshwater flow using the MEP developed stage-discharge relation. The Maraspin Creek gauge location was deemed acceptable for making flow measurements and obtaining an estimate of annual freshwater flow and attenuated nitrogen load. Calibration of the gauge was checked monthly. The gauge was installed on June 14, 2006 and was set to operate continuously for 16 months such that at least one summer season would be captured in the flow record. Stage data collection continued until December 14, 2007 for a total deployment of 18 months.

Flow in Maraspin Creek (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gauge site based upon these flow measurements and the measured water levels at the gauge site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to Barnstable Harbor - Millway and is reflective of the biological processes occurring in the small wetland up-gradient of the gauge that contributes to nitrogen attenuation (Figure IV-14 and Tables IV-5a, b and IV-6a, b). In addition, a water balance was constructed based upon the U.S. Geological Survey/MEP/CCC defined watershed delineations to determine long-term average freshwater discharge expected at the Maraspin Creek gauge site based on area and average recharge.

The annual freshwater flow record for Maraspin Creek as measured by the MEP was compared to the long-term average flows determined by the USGS/MEP/CCC modeling effort (Table III-1). The measured freshwater discharge from Maraspin Creek at the gauge location (Commercial Rd.) was nearly identical (~1% below) the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 1,384 m^3/day compared to the long term average flow determined by the watershed modeling effort of 1,396 m^3/day. The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Maraspin Creek discharging from the sub-watershed indicates that the creek is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Maraspin Creek outflow were relatively high, 2.11 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 2.93 kg/day and a measured total annual TN load of 1,051 kg/yr. Nitrate made up well more than half of the total nitrogen pool (80%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the small wetland area up-gradient of the gauge was not being transformed by plants within wetland and up-gradient subwatershed. Given the relatively high levels of remaining nitrate in the stream discharge, the Town of Barnstable should consider looking in more detail at the land uses in the subwatershed for what might be generating inorganic nitrogen load. It is possible that the agricultural activity in the subwatershed up-gradient of the gauge could be contributing through its agricultural practices. Equally important the high nitrate concentrations provide an opportunity for actions to enhance natural attenuation in this system.

From the measured nitrogen load discharged by Maraspin Creek to the upper portion of the Barnstable Harbor - Millway and the nitrogen load determined from the watershed based land use analysis, it appears that there is insignificant nitrogen attenuation of upper watershed derived nitrogen during transport to Maraspin Creek and the estuary. Based upon lower total nitrogen load (1,051 kg yr^{-1}) discharged from Maraspin Creek at the Commercial Road crossing compared to that added by the various land-uses to the associated watershed (1,360 kg yr^{-1}), there was only a 21% removal of transported nitrogen in passage through the stream and up-gradient ponds and freshwater wetlands prior to discharge to the estuary (i.e. 21% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is at the low end of the expected range of attenuation in similar watersheds. The directly measured nitrogen load from Maraspin Creek was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

Figure IV-13. Location of MEP stream gauge (yellow symbol) for measuring flow and nitrogen load in Maraspin Creek. The Maraspin Creek salt marsh receives groundwater flow from surrounding residential upland as well as a small wetland and agricultural area up-gradient of the MEP gauge. The creek discharges to the down gradient estuarine receiving waters of the Barnstable Harbor-Millway.

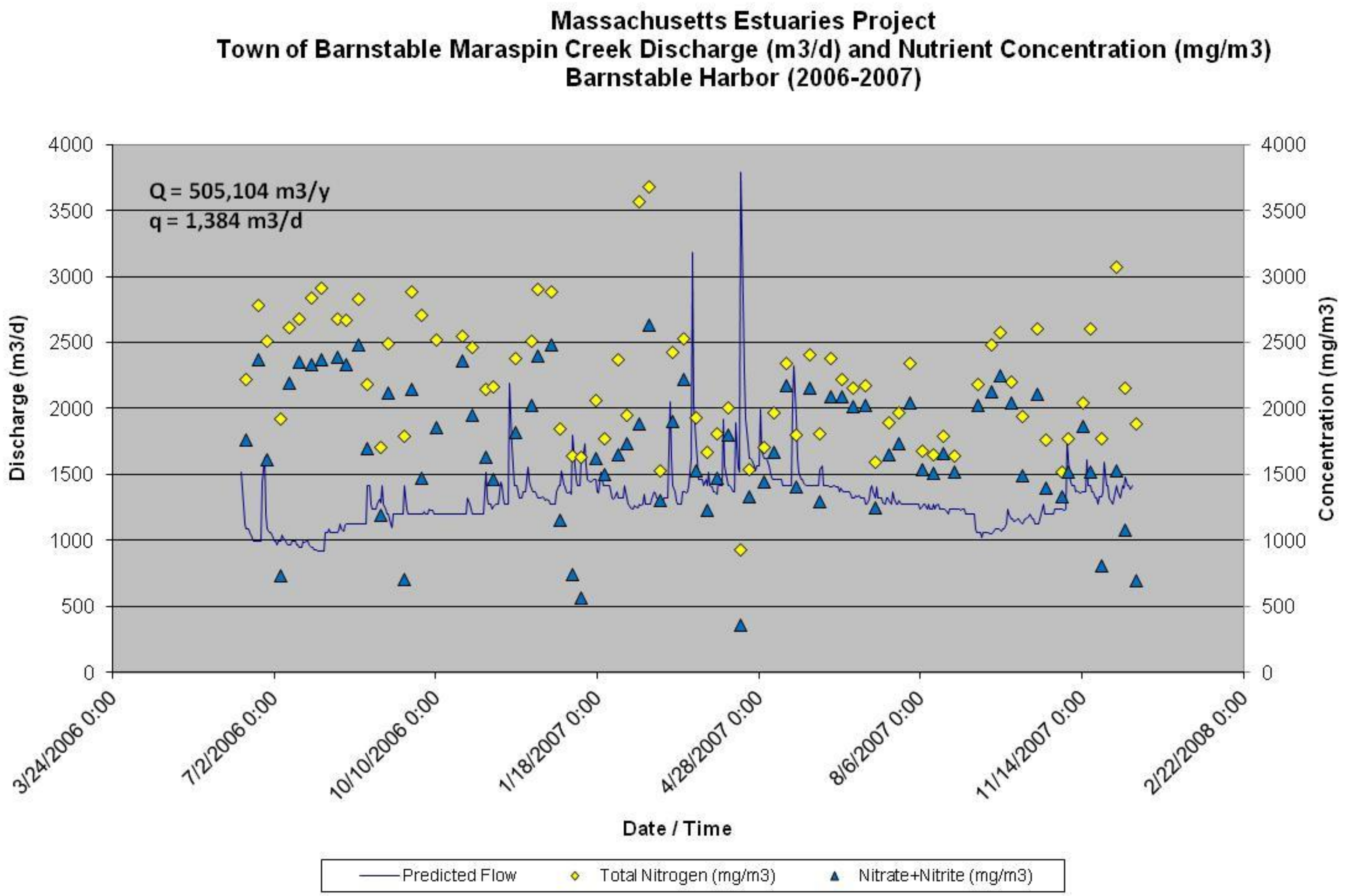


Figure IV-14. Discharge from Maraspin Creek (solid blue line). Total nitrogen (yellow symbols) and Nitrate + Nitrite (NOx, blue symbols) concentrations are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-3).

IV.2.6 Surface water Discharge and Attenuation of Watershed Nitrogen: Whites Brook Discharge to Bass Hole

Unlike most surface water features in the MEP study region, Whites Brook, which discharges into a tributary tidal creek that flows towards the main stem of Chase Garden Creek and Bass Hole, does not have an up-gradient source water pond from which that creek discharges. Rather, this small creek appears to be groundwater fed and emanates from a generally wooded area up-gradient of Route 6A (Figures IV-15a,b). The outflow leaving the wooded area up-gradient of Route 6A travels through a densely developed upland environment just prior to discharging to the head of the tidal creek proximal to Mathews Pond. The Whites Brook flow at the gauge located at the Route 6A crossing can support nitrogen attenuation to the extent that it passes through any wetlands. The combined rate of nitrogen attenuation by the biological processes occurring as the water in Whites Brook flows to the estuary was determined by comparing the present predicted nitrogen loading from the watershed contributing freshwater and associated nitrogen to the wooded areas and brook channel above the gauge site and the measured annual discharge of nitrogen to the tidal creek at the Whites Brook gauge, Figure IV-15a,b.

The freshwater flow carried by Whites Brook to the estuarine waters of Bass Hole was determined using a continuously recording vented calibrated water level gauge. As this surface water system was potentially tidally influenced, the creek discharge was checked to confirm the extent of tidal influence and whether freshwater flow could be measured at low tide in the estuary. To confirm that freshwater was being measured at low tide in the estuary, salinity measurements were conducted on weekly water quality samples collected from the gauge site. Average measured sample salinity was found to be only 0.3 ppt, clearly not tidally influenced. As such, a salinity adjustment was not necessary in order to determine daily flows using the MEP developed stage-discharge relation. The Whites Brook gauge location was deemed acceptable for making flow measurements and obtaining an estimate of annual freshwater flow and nitrogen load. Calibration of the gauge was checked monthly. The gauge was installed on June 14, 2006 and was set to operate continuously for a complete hydrologic year (low flow to low flow, ~12 months). Stage data collection continued until August 2, 2008 for a total deployment of 14 months.

Flow in the brook (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gauge site based upon these flow measurements and the measured water levels at the gauge site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge at the gauge site prior to discharge to the tributary tidal creek flowing into Bass Hole. Nitrogen removal during transport is reflective of the biological processes occurring in the channel bed and any associated wetlands or wooded areas up-gradient of the gauge at Route 6A (Figure IV-16 and Tables IV-5a, b and IV-6a, b). In addition, a water balance was constructed based upon the U.S. Geological Survey/MEP/CCC defined watershed delineations to determine long-term average freshwater discharge expected at the Whites Brook gauge site based on area and average recharge.

The annual freshwater flow record for Whites Brook as measured by the MEP was compared to the long-term average flows determined by the USGS/MEP/CCC modeling effort (Table III-1). The measured freshwater discharge from Whites Brook at the gauge location was 6% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 1,462 m^3/day compared to the long term average flows of 1,379 m^3/day determined from the watershed modeling effort. The insignificant difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Whites Brook discharging from the sub-watershed indicate that the brook is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within Whites Brook outflow were high, 0.959 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 1.40 kg/day and a measured total annual TN load of 512 kg/yr. In the Whites Brook outflow, nitrate made up slightly more than half of the total nitrogen pool (57%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the wetland areas and stream bed up-gradient of the gauge is partially taken up by and transformed to organic forms by plants within the wooded upland and the channel bed of the brook. Given the relatively high levels of remaining nitrate in the Brook waters, the possibility for additional uptake by freshwater systems appear to exist if nitrogen management actions are needed (e.g. pond or bog restoration for enhancing natural attenuation).

From the measured nitrogen load discharged by Whites Brook and the nitrogen load determined from the watershed based land use analysis, it appears that there is modest nitrogen attenuation of upper watershed derived nitrogen during transport through Whites Brook to the estuarine receiving waters. Based upon lower total nitrogen load (512 kg yr^{-1}) discharged from Whites Brook compared to that added by the various land-uses to its associated watershed, 818 kg yr^{-1}, the integrated attenuation in passage through this small brook and associated aquatic components is 37% (i.e. 37% of nitrogen input to watershed does not reach the estuary). This level of attenuation is consistent with other streams/creeks/brooks evaluated under the MEP (including to the Barnstable Great Marshes). The directly measured nitrogen load from Whites Brook was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

Figure IV-15a. Location of two MEP stream gauges (yellow symbol) on streams discharging to the Bass Hole sub-embayment (southern stream is Whites Brook and northern stream is Chase Garden Creek). Freshwater flow and nitrogen load were measured over a complete year at each gauge site. Bass Hole is a major tributary to the Barnstable Great Marsh estuary system.

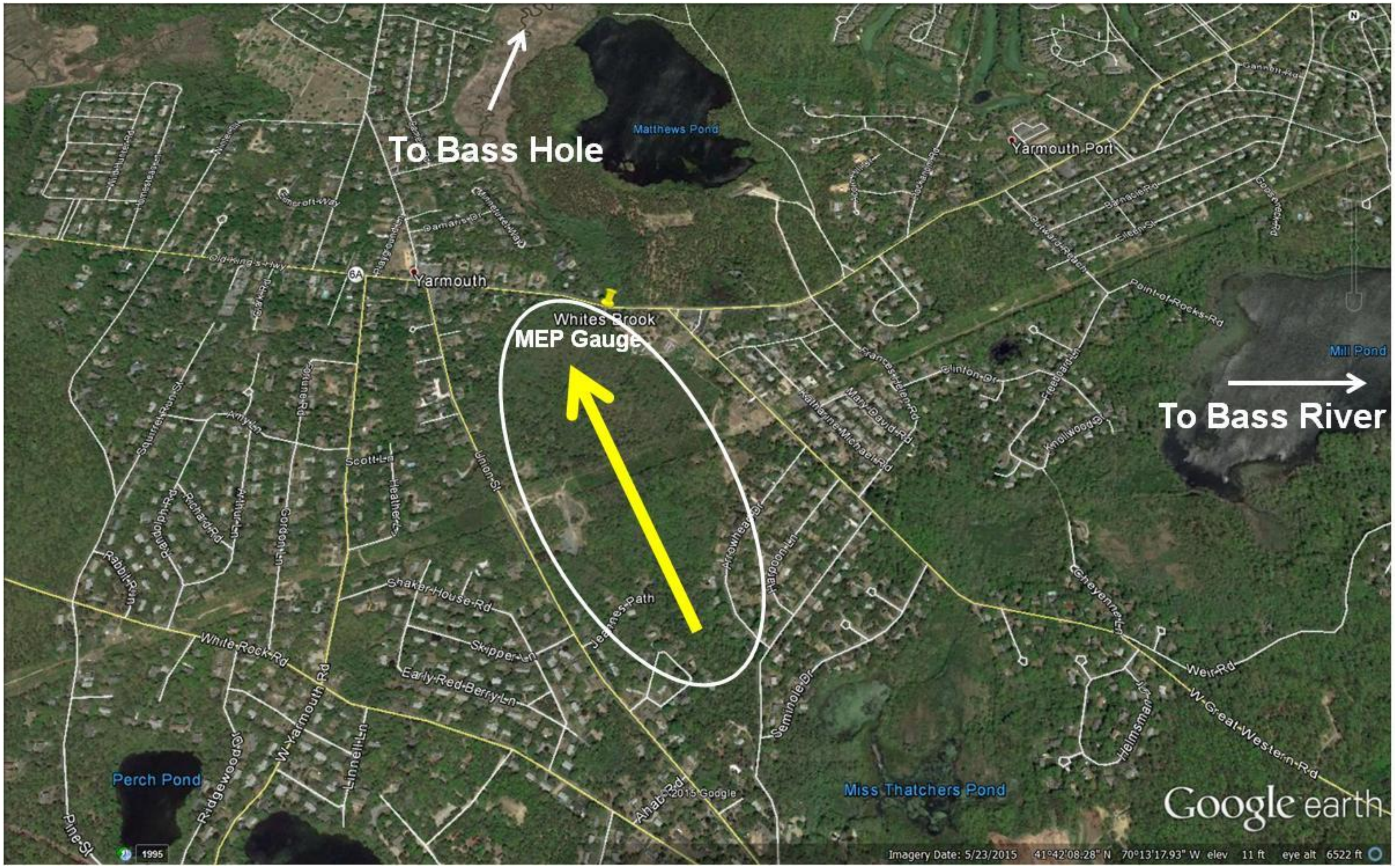


Figure IV-15b. Location of MEP stream gauge (yellow symbol) for measuring flow and nitrogen load in Whites Brook. Yellow arrows show the direction of surfacewater flow draining a small wooded area up-gradient of the Whites Brook gauge.

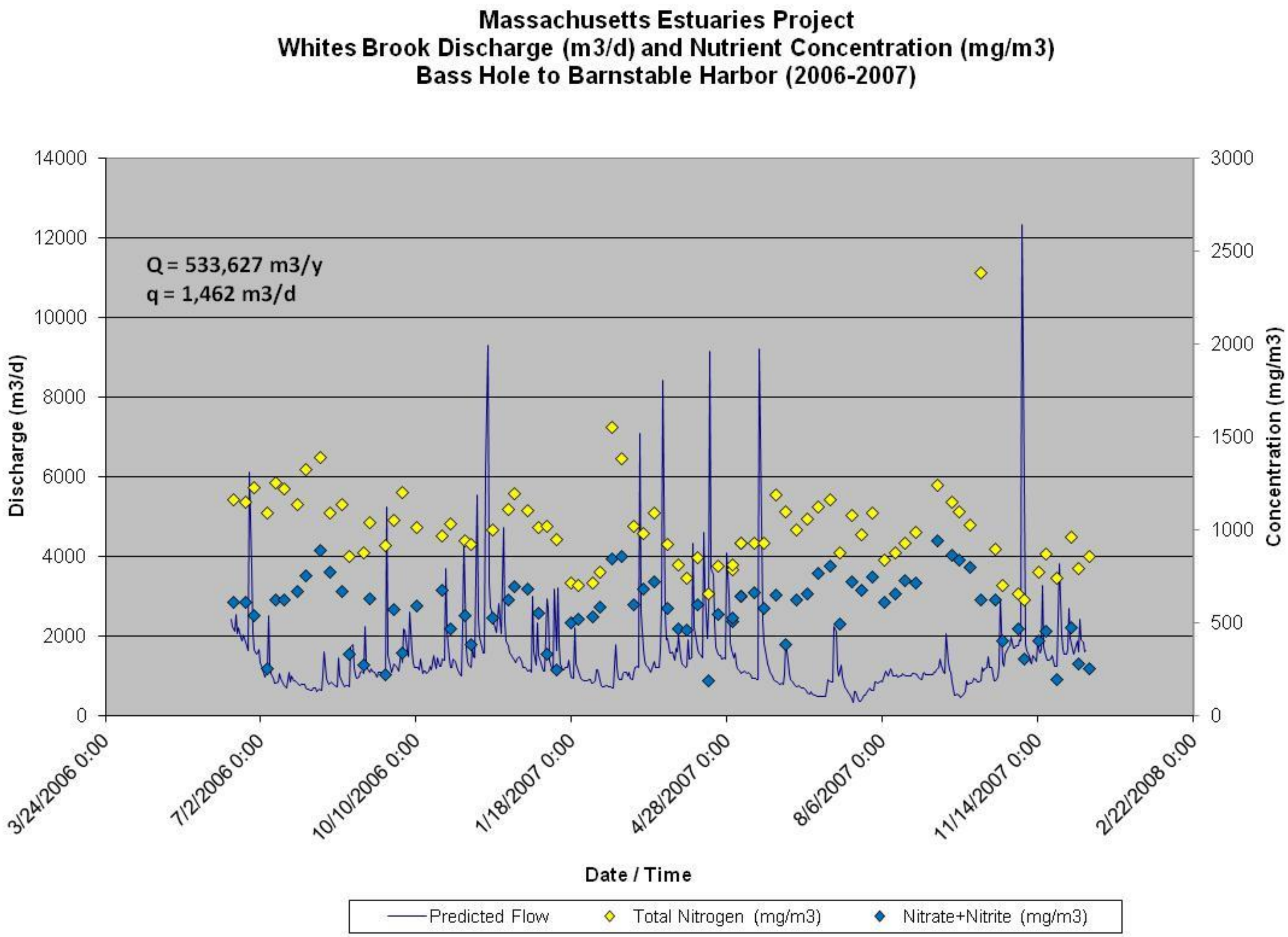


Figure IV-16. Discharge from Whites Brook to Bass Hole (solid blue line). Total nitrogen (yellow symbols) and Nitrate + Nitrite (NOx, blue symbols) concentrations are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-5a, b).

IV.2.7 Surface water Discharge and Attenuation of Watershed Nitrogen: Chase Garden Creek Discharge to Bass Hole

Unlike most surface water features in the MEP study region that typically emanate from a specific pond or wetland, Chase Garden Creek, which discharges directly into Bass Hole, does not have an upgradient sourcewater pond from which that Creek discharges. Rather, the headwaters of the creek emanate from what appear to be bogs situated proximal to the shoreline of Cape Cod Bay, with much of the creek being groundwater fed as it flows through a generally residential area with the area upgradient of the MEP gauge being generally marshy with houses to either side of the creek. This marshy area upgradient of the gauge located at the New Boston Road crossing may be providing some attenuation of nitrogen. The culvert location at New Boston Road provides a good point for a direct measurement of flow and nitrogen load in the creek. The combined rate of nitrogen attenuation by the biological processes occurring as the water in Chase Garden Creek flows toward the estuary was determined by comparing the present predicted nitrogen loading to the sub-watershed region up-gradient of the MEP gauge and the measured annual discharge of nitrogen to Bass Hole at the Chase Garden Creek gauge, Figures IV-15a and 17.

The freshwater flow carried by Chase Garden Creek to the estuarine waters of Bass Hole was determined using a continuously recording vented calibrated water level gauge. As this surface water system was potentially tidally influenced, the creek discharge was checked to confirm the extent of tidal influence and whether freshwater flow could be measured at low tide in the estuary. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average measured sample salinity was found to be 0.3 ppt, clearly not tidally influenced. As such, a salinity adjustment was not necessary in order to determine daily flows using the MEP developed stage-discharge relation. The Chase Garden Creek gauge location was deemed acceptable for making flow measurements and obtaining an estimate of annual freshwater flow to the estuary. Calibration of the gauge was checked monthly. The gauge was installed on June 14, 2006 and was set to operate continuously for a complete hydrologic year (low flow to low flow, ~12 months). Stage data collection continued until August 2, 2007 for a total deployment of 15 months.

Flow in the creek (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gauge site based upon these flow measurements and the measured water levels. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the Bass Hole sub-estuary and integrates both watershed nitrogen inputs and nitrogen removals during transport by biological processes occurring in the channel bed and any associated wetlands (Figure IV-18 and Tables IV-5a, b and IV-6a, b). In addition, a water balance was constructed based upon the U.S. Geological Survey/MEP/CCC defined watershed delineations to determine long-term average freshwater discharge expected at the Chase Garden Creek gauge site based on contributing area and average recharge.

The annual freshwater flow record for Chase Garden Creek as measured by the MEP was compared to the long-term average flows determined by the USGS/MEP/CCC modeling effort (Table III-1). The measured freshwater discharge from Chase Garden Creek at the gauge location was only 2% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 1,611 m^3/day compared to the long term average flows determined by the watershed modeling effort (1,574 m^3/day). The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Chase Garden Creek indicates that the creek is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Chase Garden Creek outflow were high, 1.05 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 1.70 kg/day and a measured total annual TN load of 622 kg/yr. In the Chase Garden Creek outflow, nitrate made up less than half of the total nitrogen pool (35%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the wetland areas and stream bed up-gradient of the gauge is only partially taken up and transformed by plants to organic forms within the riparian zone and the channel bed of the creek. Given the moderate levels of remaining nitrate in the discharge water from the creek, the possibility for additional uptake by freshwater systems may be limited in the Chase Garden Creek sub-watershed.

From the measured nitrogen load discharged by Chase Garden Creek to Bass Hole and the nitrogen load determined from the watershed based land use analysis, it appears that there is significant nitrogen attenuation of upper watershed derived nitrogen during transport to the Chase Garden Creek gauge location and the estuarine receiving waters. Based upon lower total nitrogen load (622 kg yr^{-1}) discharged from Chase Garden Creek at New Boston Road compared to that added by the various land-uses to the associated watershed (1,561 kg yr^{-1}), the integrated attenuation in passage through this Creek and the marshy riparian areas and bogs is relatively high at 60% (i.e. 60% of nitrogen input to watershed does not reach the estuary). This level of attenuation is consistent with other streams/creeks/brooks evaluated under the MEP with transport through areas with marshy creek banks and wetlands/bogs which are capable of attenuating nitrogen. The directly measured nitrogen load from Chase Garden Creek was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

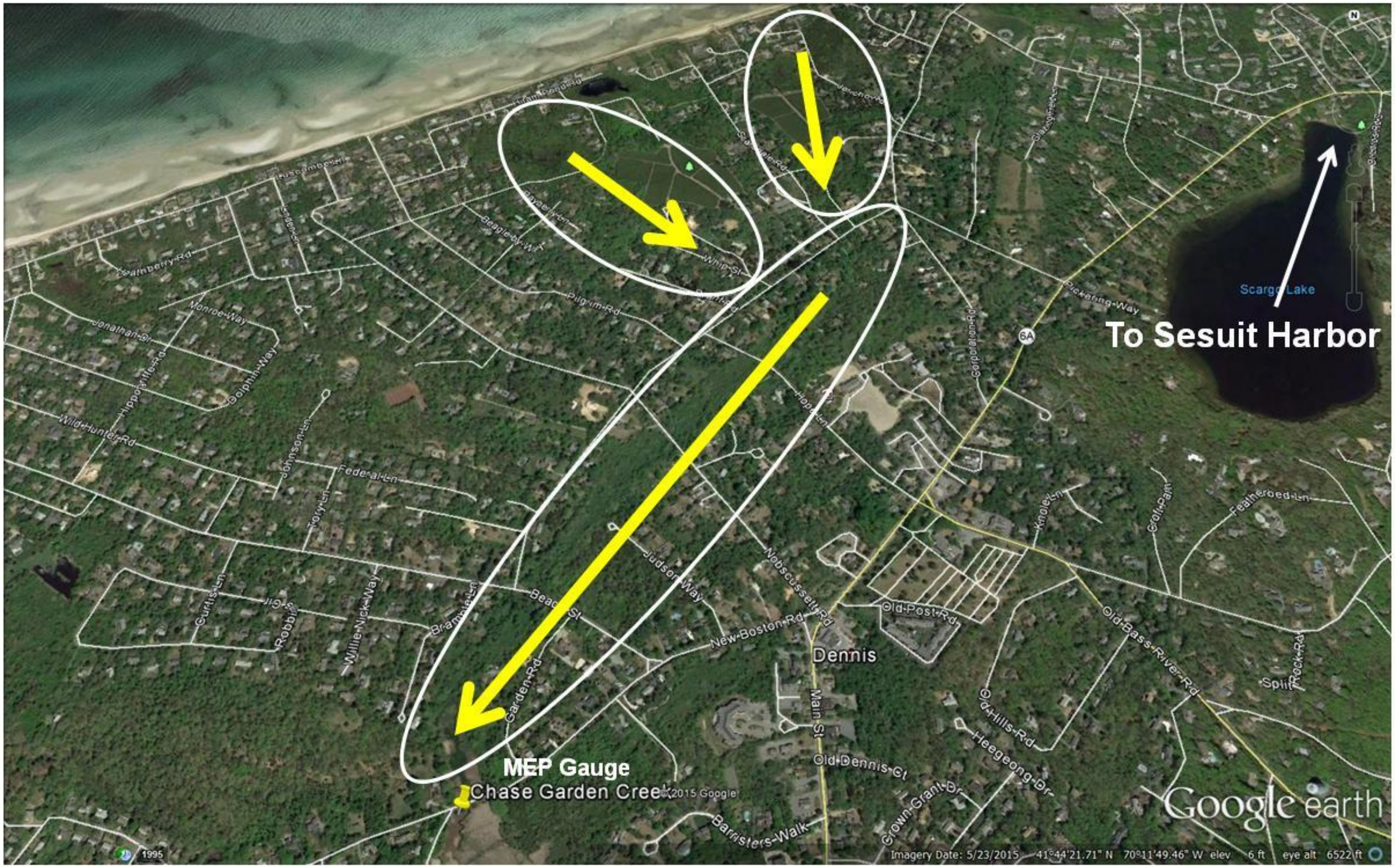


Figure IV-17. Location of MEP stream gauge (yellow symbol) for measuring flow and nitrogen load in Whites Brook. Yellow arrows show the direction of surfacewater flow draining a marshy riparian zone along Chase Garden Creek as well as a network of bogs as the head waters of the Creek.

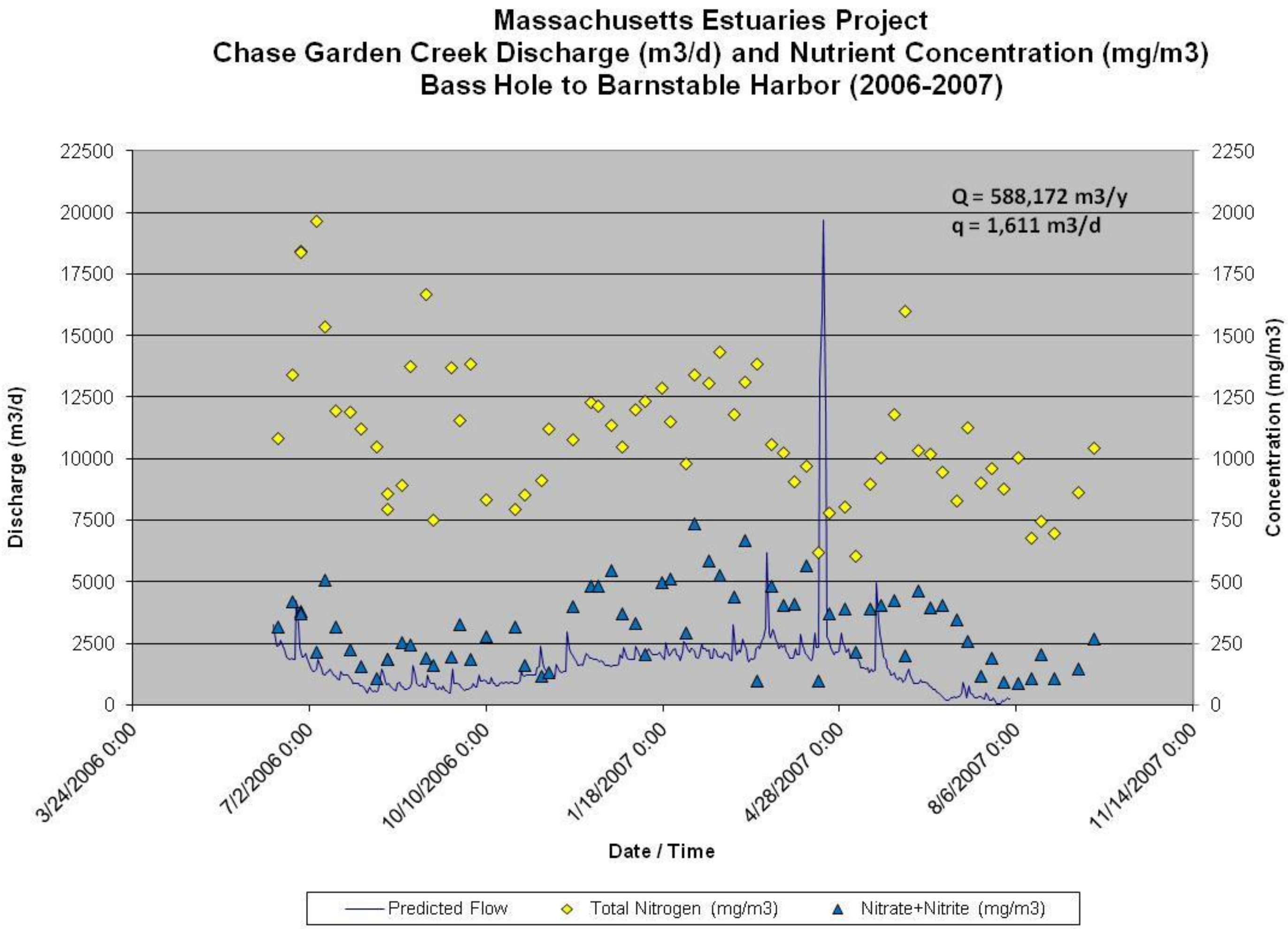


Figure IV-18. Discharge from Chase Garden Creek to Bass Hole (solid blue line). Total nitrogen (yellow symbols) and Nitrate + Nitrite (NOx, blue symbols) concentrations are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-5a, b).

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

The overall objective of the benthic nutrient flux survey of the Barnstable Great Marsh estuary system was to quantify the summertime exchange of nitrogen, between the sediments and overlying waters throughout this large system. The mass exchange of nitrogen between water column and sediments is a fundamental factor in controlling nitrogen levels within coastal waters. These fluxes and their associated biogeochemical pools relate directly to carbon, nutrient and oxygen dynamics and the nutrient related ecological health of these shallow marine ecosystems. In addition, these data are required for the proper modeling of nitrogen in shallow aquatic systems, both fresh and salt water.

IV.3.1 Sediment-Water column Exchange of Nitrogen

As stated in the above section, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the Barnstable Great Marsh estuary system predominantly in highly bio-available forms from the surrounding upland watersheds and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the water column (once it entered) then predicting water column nitrogen levels would be simply a matter of determining the watershed loads, dispersion, and hydrodynamic flushing. However, as nitrogen enters the embayment from the surrounding watersheds it is predominantly in the bio-available form nitrate. This nitrate and other bio-available forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton “particles”. Most of these “particles” remain in the water column for sufficient time to be flushed out to a down gradient larger water body (like Cape Cod Bay). However, some of these phytoplankton particles are grazed by zooplankton or filtered from the water by shellfish and other benthic animals and deposited on the bottom. Also, in longer residence time systems (greater than 8 days) these nitrogen rich particles may die and settle to the bottom. In both cases (grazing or senescence), a fraction of the phytoplankton with their associated nitrogen “load” become incorporated into the surficial sediments of the embayment.

In general the fraction of the phytoplankton population which enters the surficial sediments of a shallow embayment: (1) increases with decreased hydrodynamic flushing, (2) increases in low velocity settings, (3) increases within enclosed tributary basins, particularly if they are deeper than the adjacent embayment. To some extent, the settling characteristics can be evaluated by observation of the grain-size and organic content of sediments within an estuary.

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bio-available nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by SMAST and the MEP, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. In contrast in some systems, with deep depositional basins or salt marsh tidal creeks, the sediments can be a net sink for nitrogen even during summer (e.g. Namskaket Salt Marsh (lower reach), Mashapaquit Creek Salt Marsh, West Falmouth Harbor; Centerville River Salt Marsh or Sesachacha Pond on the Island of Nantucket). Embayment basins can also be net sinks for nitrogen to the extent that they support relatively oxidized surficial sediments, for example in the margins of the main basin to Lewis Bay (Town of Barnstable, Cape Cod). In contrast, most embayments show low rates of nitrogen release throughout much of their basin areas and, in regions of high deposition the anoxic sediments show high release rates during summer months, due to a lessening of coupled nitrification-denitrification. The consequence of high deposition rates is that the basin sediments are unconsolidated, organic rich and sulfidic nature (MEP field observations).

Failure to account for the site-specific nitrogen balance of the sediments and its spatial variation from the tidal creeks and embayment basins will result in significant errors in determination of the threshold nitrogen loading to the Barnstable Great Marsh estuary system. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for determining sediment-watercolumn nitrogen exchange

For the Barnstable Great Marsh estuary system, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from a total of 40 cores from 38 sites in the Great Marshes, Barnstable Harbor and Bass Hole portions of the overall system. All the sediment cores for this system were collected in July-August 2006. Measurements of total dissolved nitrogen, nitrate + nitrite (NOx), ammonium were made in time-series on each incubated core sample.

Rates of nitrogen release were determined using undisturbed sediment cores incubated for 24 hours in temperature-controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by small boat to a shore side lab. Cores were maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from core sites to replace the headspace water of each core prior to incubation. The number of core samples from each estuarine component (Figure IV-19) are as follows:

Barnstable Harbor Benthic Nutrient Regeneration Cores

- BH-1 1 core (Scorton Creek)
- BH-2 1 core (Scorton Creek)
- BH-3 1 core (Scorton Creek)
- BH-4 1 core (Scorton Water)
- BH-5 1 core (Spring Creek)
- BH-6 1 core (Spring Creek)
- BH-7 1 core (Spring Creek)
- BH-8 1 core (Spring Creek)
- BH-9 1 core (Main Basin Upper)
- BH-10 1 core (Brickyard Creek)
- BH-11 1 core (Brickyard Creek)
- BH-12 1 core (Brickyard Creek)
- BH-13 1 core (Brickyard Creek)
- BH-14 1 core (Brickyard Creek)
- BH-15 1 core (Main Basin Upper)
- BH-16 1 core (Main Basin Upper)

• BH-17	1 core	(Main Basin Upper)
• BH-18	1 core	(Main Basin Upper)
• BH-19	1 core	(Main Basin Upper)
• BH-20 + 21	2 cores	(Main Basin Upper)
• BH-22	1 core	(Main Basin Upper)
• BH-23	1 core	(Millway)
• BH-24 + FD	2 cores	(Millway)
• BH-25	1 core	(Millway)
• BH-26	1 core	(Main Basin Lower)
• BH-27	1 core	(Main Basin Lower)
• BH-28	1 core	(Main Basin Lower)
• BH-29	1 core	(Wharf Creek)
• BH-30	1 core	(Wharf Creek)
• BH-31	1 core	(Lone Tree Creek)
• BH-32	1 core	(Lone Tree Creek)
• BH-33	1 core	(Lone Tree Creek)
• BH-34	1 core	(Lone Tree Creek)

Bass Hole Benthic Nutrient Regeneration Cores

• BASS-1	1 core	(Bass Hole Lower)
• BASS-2	1 core	(Bass Hole Lower)
• BASS-3/4	2 cores	(Bass Hole Lower)
• BASS-5	1 core	(Bass Hole Upper)
• BASS-6	1 core	(Bass Hole Upper)

Sediment-water column exchange follows the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1998) for nutrients and metabolism. Upon return to the field laboratory at the Millway Marina facility on the shore of Barnstable Harbor, the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and ortho-phosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining samples frozen (-20^{o}C) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

Chemical analyses were performed by the Coastal Systems Analytical Facility at the School for Marine Science and Technology (SMAST) at the University of Massachusetts in New Bedford, MA. (508-910-6325 or d1white@umassd.edu). The laboratory follows standard methods for saltwater analysis and sediment geochemistry.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

Water column nitrogen levels are the balance of inputs from direct sources (land, rain etc), losses (denitrification, burial), regeneration (water column and benthic), and uptake (e.g.

photosynthesis). As stated above, during the warmer summer months the sediments of shallow estuaries typically act as a net source of nitrogen to the overlying waters and help to stimulate eutrophication in organic rich systems. However, some sediments may be net sinks for nitrogen and some may be in "balance" (organic N particle settling = nitrogen release). Sediments may also take up dissolved nitrate directly from the water column and convert it to dinitrogen gas (termed "denitrification"), hence effectively removing it from the ecosystem. This process is typically a small component of sediment denitrification in embayment sediments, since the water column nitrogen pool is typically dominated by organic forms of nitrogen, with very low nitrate concentrations. However, this process can be very effective in removing nitrogen loads in some systems, particularly in streams, ponds and salt marshes, where overlying waters support high nitrate levels.

In addition to nitrogen cycling, there are ecological consequences to habitat quality of organic matter settling and mineralization within sediments, these relate primarily to sediment and water column oxygen status. However, for the modeling of nitrogen within an embayment it is the relative balance of nitrogen input from water column to sediment versus regeneration which is critical. Similarly, it is the net balance of nitrogen fluxes between water column and sediments during the modeling period that must be quantified. For example, a net input to the sediments represents an effective lowering of the nitrogen loading to down-gradient systems and net output from the sediments represents an additional load.

The relative balance of nitrogen fluxes ("in" versus "out" of sediments) is dominated by the rate of particulate settling (in), the rate of denitrification of nitrate from overlying water (in), and regeneration (out). The rate of denitrification is controlled by the levels of organic matter within the sediments, whether the sediments are oxic or anoxic and the concentration of nitrate in the overlying water. Organic rich sediment systems with high overlying nitrate frequently show large net nitrogen uptake throughout the summer months, even though organic nitrogen is being mineralized and released to the overlying water as well. The rate of nitrate uptake, simply dominates the overall sediment nitrogen cycle.

In order to model the nitrogen distribution within an embayment it is important to be able to account for the net nitrogen flux from the sediments within each part of each system. This requires that an estimate of the particulate input and nitrate uptake be obtained for comparison to the rate of nitrogen release. Only sediments with a net release of nitrogen contribute a true additional nitrogen load to the overlying waters, while those with a net input to the sediments serve as an "in embayment" attenuation mechanism for nitrogen, lowering nitrogen levels.

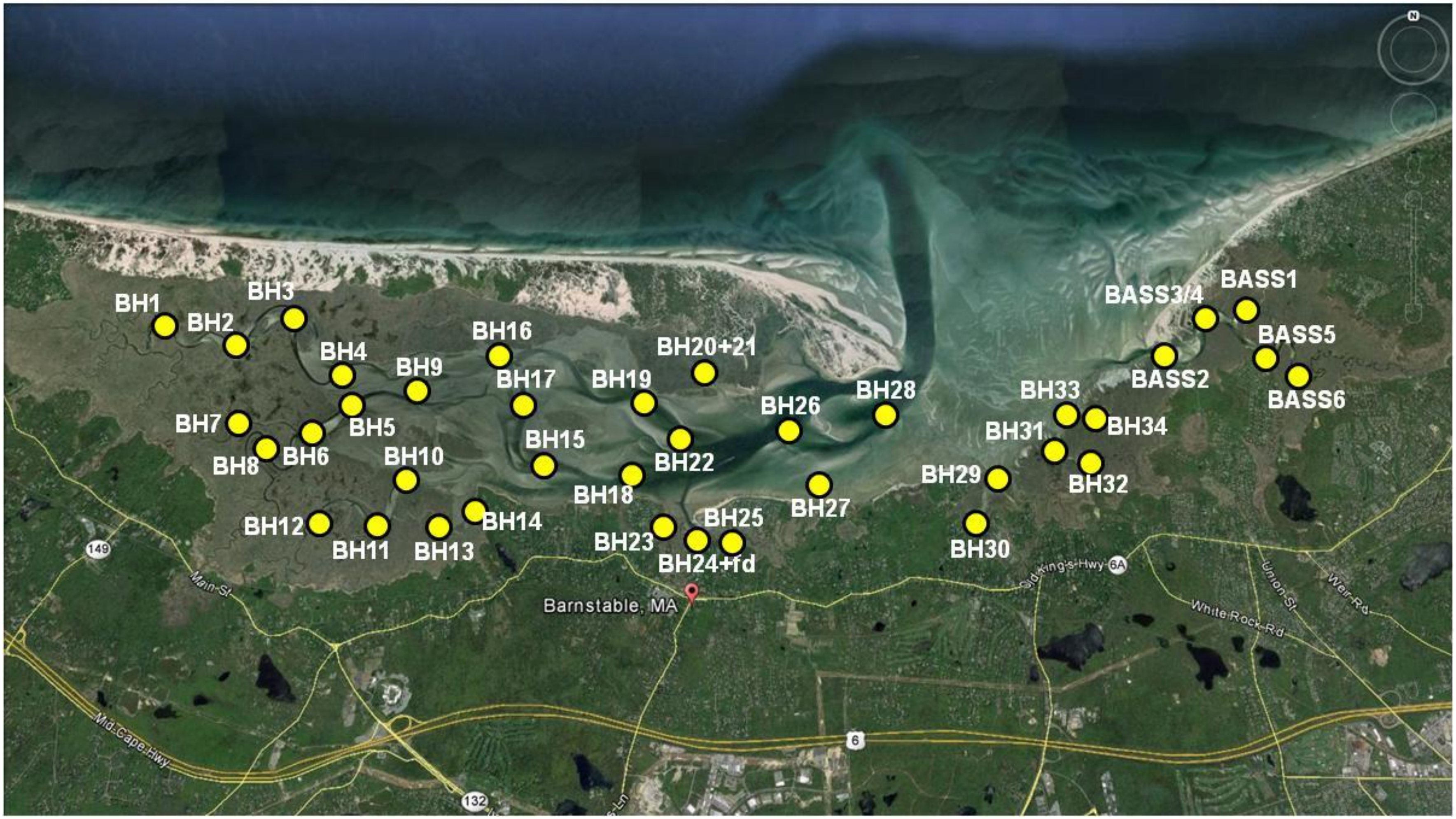


Figure IV-19. Barnstable Great Marsh estuary system sediment sampling sites (yellow symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

Overall, coastal sediments are not overlain by nitrate rich waters and the major nitrogen input is via phytoplankton grazing or direct settling. In these systems, on an annual basis, the amount of nitrogen input to sediments is generally higher than the amount of nitrogen release. This net sink results from the burial of reworked refractory organic compounds, sorption of inorganic nitrogen and some denitrification of produced inorganic nitrogen before it can "escape" to the overlying waters. However, this net sink evaluation of coastal sediments is based upon annual fluxes. If seasonality is taken into account, it is clear that sediments undergo periods of net input and net output. The net output is generally during warmer periods and the net input is during colder periods. The result can be an accumulation of nitrogen within late fall, winter, and early spring and a net release during summer. The conceptual model of this seasonality has the sediments acting as a battery with the flux balance controlled by temperature (Figure IV-20).

Unfortunately, the tendency for net release of nitrogen during warmer periods coincides with the periods of lowest nutrient related water quality within temperate embayments. This sediment nitrogen release is in part responsible for poor summer nutrient related health. Other major factors causing the seasonal water quality decline are the lower solubility of oxygen during summer, the higher oxygen demand by marine communities, and environmental conditions supportive of high phytoplankton growth rates.

In order to determine the net nitrogen flux between water column and sediments, all of the above factors were taken into account. The net input or release of nitrogen within a specific embayment was determined based upon the measured total dissolved nitrogen uptake or release, and estimate of particulate nitrogen input.

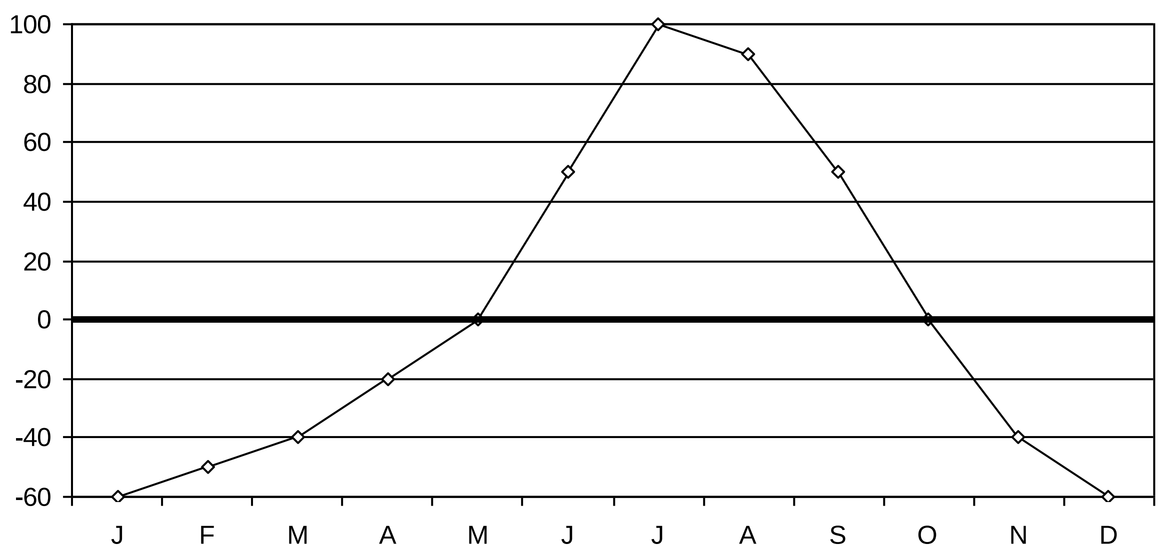


Figure IV-20. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

Sediment Nitrogen Release by Standard Core Approach: Sediment sampling was conducted throughout the main tidal channels of the salt marsh portions of the Barnstable Great Marsh estuary system as well as the open water main of the estuary. The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content, as well as sediment type and an analysis of each site's tidal flow velocities. As expected flow velocities are generally low in the uppermost reaches of the tidal creeks and high in the lower portions of the system situated closer to the inlet to the system. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site, the average summer particulate carbon and nitrogen concentration within the overlying water and the tidal velocities from the hydrodynamic model (Section V). Generally two levels of settling are used. If the sediments were organic rich and fine grained, and the hydrodynamic data showed low tidal velocities, then a water column particle residence time of 8 days was used (based upon phytoplankton and particulate carbon studies of poorly flushed basins). If the sediments indicated coarse-grained sediments and low organic content and high velocities, then half this settling rate was used. However, in the relatively small areas of very high velocity near inlets or main tidal channels or areas of swept sands, a further reduction in deposition is applied. Adjusting the measured sediment releases was essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas which are net nitrogen sinks for the aquatic system. This approach has been previously validated in outer Cape Cod embayments (Town of Chatham embayments) by examining the relative fraction of the sediment carbon turnover (total sediment metabolism), which would be accounted for by daily particulate carbon settling. This analysis indicated that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins is driven primarily by stored organic matter (ca. 90%). Also, in the more open lower portions of larger embayments, storage appears to be low and a large proportion of the daily carbon requirement in summer is met by particle settling (approximately 33% to 67%). This range of values and their distribution is consistent with ecological theory and field data from shallow embayments. Additional, validation has been conducted on deep enclosed basins (with little freshwater inflow), where the fluxes can be determined by multiple methods. In this case the rate of sediment regeneration determined from incubations was comparable to that determined from whole system balance.

Net nitrogen release or uptake from the sediments within the Barnstable Great Marsh estuary system were comparable to other salt marsh dominated systems on Cape Cod with similar configurations and flushing rates (e.g. Wellfleet Harbor, Scorton Creek, Sandwich Harbor, Namskaket Marsh, Little Namskaket Marsh). The spatial distribution of nitrogen release/uptake by the sediments within the Barnstable Great Marsh estuary system showed a clear pattern associated with sediment type, water velocity and the associated depositional environments found within the component basins of the estuary. Net nitrogen release was highest (72.9 mg N m^{-2} d^{-1}) in the artificially deepened basins of Barnstable Harbor (millway) which is deepened for navigation which creates ideal conditions for deposition. The organic rich tidal marsh creeks showed low nitrogen release rates, 0.2 – 3.5 mg N m^{-2} d^{-1}, while the high velocity sandy areas in the central main basin of the Great Marshes and the inlet to Bass Hole supported net nitrogen uptake, -3.4 to -6.1 mg N m^{-2} d^{-1}. This pattern of sediment nitrogen release has been frequently observed in other salt marsh dominated systems assessed by the MEP. For example, the adjacent Scorton Creek Estuary showed net nitrogen release in the organic rich upper tidal reaches, 25.5 mg N m^{-2} d^{-1} to net uptake throughout the lower sandy tributary creeks to Long Hill and Scorton Shore, -19.9 and -12.7 mg N m^{-2} d^{-1}, respectively, with the high velocity larger tidal creeks with little organic matter accumulation showing low net uptake, -3.0 mg N m^{-2} d^{-1}. The same pattern was observed in another Cape Cod Bay salt marsh, Namskaket Creek, where the upper marsh creeks also showed a net release, 45.4 mg N m^{-2} d^{-1}, and the lower creek areas net uptake, -21.2 mg N m^{-2} d^{-1}. Again in Little Namskaket Marsh, the upper marsh creeks showed a net release, 64.5 mg N m^{-2} d^{-1}, and the lower creek areas net uptake, -7.8 mg N m^{-2} d^{-1}.

Within the Barnstable Great Marsh estuary system rates of uptake were also similar to other salt marsh systems on Cape Cod with lower tide ranges than the 10 ft range in marshes tributary to Cape Cod Bay. For example, net nitrogen uptake in the high velocity sandy areas (-3.4 to -6.1 mg N m^{-2} d^{-1}) was similar to that observed for the salt marsh areas in the Centerville River System (-4.5 to -13.2 mg N m^{-1} d^{-1}) and Cockle Cove Salt Marsh, Chatham (MEP Centerville River Final Nutrient Technical Report 2006, MEP Cockle Cove Technical Memorandum-Howes et al. 2006) and the lower basin of the Little River marsh system (-3.1 mg N m^{-1} d^{-1}) on Buzzards Bay. The net release rates from the upper tidal reaches are similar and comparable to other systems. For example, the upper reaches of the Herring River wetland system (9.7-10.5 mg N m^{-1} d^{-1}), Wild Harbor River (1.4 mg N m^{-1} d^{-1}) and salt marsh dominated portions of the Back River (Bourne) and the Slocums and Little River Estuaries (Dartmouth) support similarly small net release rates of 6.5 mg N m^{-2} d^{-1} and 4.6-9.0 mg N m^{-2} d^{-1}, respectively.

Net nitrogen release rates for use in the water quality modeling effort for the component sub-basins of the Barnstable Great Marsh estuary system (Section VI) are presented in Table IV-7 There was a clear spatial pattern of sediment nitrogen flux, with net release from the sediments of the upper reach of the tidal creek and net uptake by the sediments of the lower tidal creek region. The magnitude and pattern of sediment nitrogen release with the Barnstable Great Marsh estuary system is consistent with the distribution of sediment types and deposition rates and is consistent with other similarly structured salt marsh dominated estuaries with low to moderate watershed nitrogen loading. Equally important, sediment processes appear to be in balance with the overlying waters and the nitrogen flux rates consistent with the low nitrogen loading to this system and it relatively high flushing rate.

Table IV-7. Rates of net nitrogen return from sediments to the overlying waters of component basins comprising the Barnstable Great Marsh estuary system. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (Section VI). Measurements represent July -August rates.

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			Station I.D. *
	Mean	S.E.	# sites	Sta-#
Barnstable Great Marsh estuary system				
Bass Hole Estuary				
Upper	**0.2**	**4.0**	**2**	**Bass 5-6**
Lower	**-6.1**	**1.4**	**4**	**Bass 1-4**
Barnstable Great Marshes				
Central Main Basin Upper	**-5.4**	**2.2**	**4**	**BH 9, 15-17**
Central Main Basin Lower	**-3.4**	**10.3**	**8**	**BH 18-22, 26-28**
Scorton Creek	**0.6**	**13.1**	**4**	**BH 1-4**
Spring Creek	**2.6**	**6.9**	**4**	**BH 5-8**
Brickyard Creek	**2.5**	**13.8**	**5**	**BH 10-14**
Barnstable Harbor	**72.9**	**31.1**	**5**	**BH 23-25**
Wharf Creek	**3.5**	**0.4**	**2**	**BH 29-30**
Lone Tree Creek	**2.8**	**5.1**	**4**	**BH 31-34**

* Station numbers refer to Figure IV-19.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This hydrodynamic study was performed for Barnstable Harbor, located on the Cape Cod Bay facing shoreline of Barnstable and Yarmouth, Massachusetts to characterize flow and circulation into and out of the system and serves as the basis for nutrient related water quality modeling discussed in Section VI. Barnstable Harbor is the receiving basin of groundwater flow from the historic village of Barnstable, the county seat. A topographic map detail in Figure V-1 shows the general study area. As modeled, the system is comprised of a broad marsh plain called the “Great Marshes” (van der Molen, 1997) at the western end, the main harbor basin in the middle of the system, and another large area of salt marsh at the eastern end, named Chase Garden Creek, which includes the area referred to as Bass Hole. A portion of Chase Garden Creek lies within the Town of Dennis. The Great Marsh and Main Harbor basin are bound to the north by Sandy Neck which extends between the Sandwich town border and the Harbor inlet. The lowest elevations of the system exist in the main natural channel, where maximum depths of approximately -36 feet NAVD occur near the eastern tip of Sandy Neck. The surface coverage of the Barnstable Harbor and Chase Garden Creek together is more than 7,900 total acres, which includes more than 4,400 acres of marsh plain

Tidal exchange with Cape Cod Bay dominates circulation in the Harbor. From measurements made in the course of this study, the average offshore tide range is 10 feet. As the tide propagates through the marsh, its range is attenuated. At interior marsh creek gauge stations deployed for this study, the range was measured to be 6 feet. The reduction in the tide range is mostly due to a truncation of the lower half of the tide, which is caused by the elevation of the marsh creeks in the upper reaches of the marsh plain of the Harbor.

The hydrodynamic study of the Barnstable Harbor and Chase Garden Creek system proceeded as two main efforts that dealt with data collection and development of the hydrodynamic model. In the first portion of the study, bathymetry and tide data were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry survey of Barnstable Harbor and Chase Garden Creek was performed to determine the variation of depths throughout the main sub-embayment areas of the system. This survey addressed the previous lack of adequate bathymetry data for these areas. In addition to the bathymetry survey, tides were recorded at five stations within the Harbor system for 31 days. These tide data were necessary to run and calibrate the hydrodynamic model of the system.

A numerical hydrodynamic model of Barnstable Harbor and its attached sub-embayments was developed in the second portion of this study. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from Cape Cod Bay were used to define the open boundary condition that drives the circulation of the model. Data measured within the system were used to calibrate and verify model performance to ensure that it accurately represents the dynamics of the real, physical system.

The calibrated hydrodynamic model of Barnstable Harbor is an integral piece of the water quality model developed in Section 6 this MEP nutrient threshold report. In addition to its use as the hydrodynamic basis for the TN and salinity models, the calibrated hydrodynamic model is a useful tool that can be used to investigate the tidal properties of the system.

V.2 DATA COLLECTION AND ANALYSIS

The field data collection portion of this study was performed to characterize the physical properties of Barnstable Harbor. Bathymetry data were collected throughout the system so that it could be accurately represented as a computer hydrodynamic model and flushing rates could be determined for the system. In addition to the bathymetry, tide data were also collected throughout the Harbor in order to run the circulation model with real tides, and also to calibrate and verify model performance.

Figure V-1. Topographic map detail of Barnstable Harbor and Chase Garden Creek.

V.2.1 Bathymetry Data

Bathymetric data was collected in the Harbor (including the main marsh creeks) during the summer of 2015. The survey employed a boat-mounted fathometer. Positioning data were collected using a differential GPS. Where practical, predetermined survey transects were followed at regular intervals. Collected bathymetry data was tide-corrected to account for the change in water depths as the tide level changed over the survey period. The tide-correction is performed using tide data collected while the survey was run. For the broad areas of tide flats and marsh plain not covered in the boat survey, elevation data were available from a 2013 USGS LiDAR flight of the area. The complied elevation dataset, including all sources of bathymetry and topography, is shown in Figure V-2.

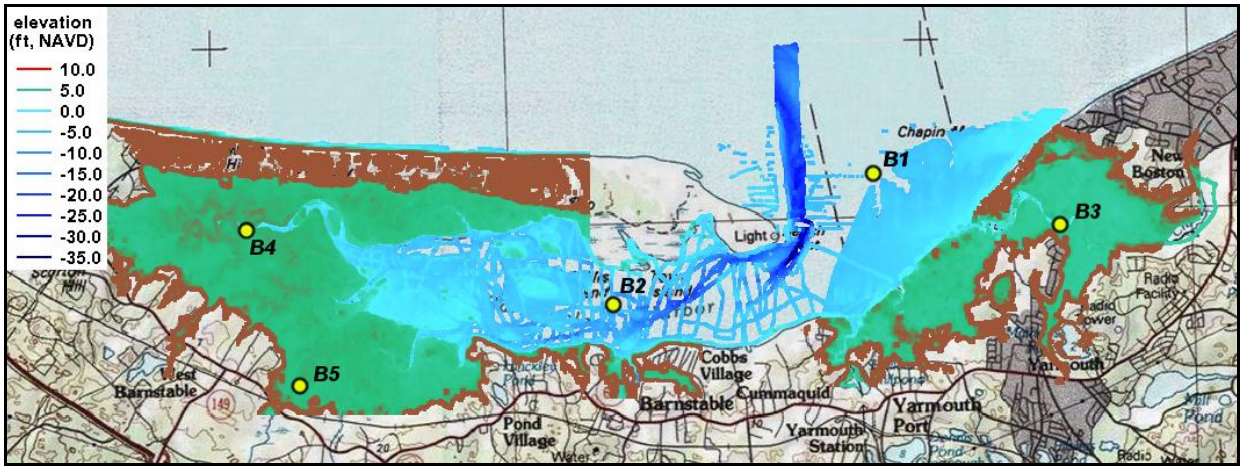


Figure V-2. Bathymetric and topography data used to develop the RMA-2 hydrodynamic model. Points are colored to represent the elevation relative to NAVD. The data sources used to develop the grid mesh are the 2015 bathymetry survey conducted by the MEP Technical Team and USGS 2013 LiDAR topography. Location of tide gauges are also indicated.

V.2.2 Tide Data Collection and Analysis

Tide data records were collected concurrently at five gauging stations shown in Figure V-2, located at the harbor opening to Cape Cod Bay (B1), in the main basin of the harbor (B2), at Chase Garden Creek (B3), in Scorton Creek Cove (B4) and in Bridge Creek (B5). The Temperature Depth Recorders (TDR) used to record the tide data were deployed for a 51-day period between June 16 and August 6, 2015. The elevation of each gauge was surveyed relative to the NAVD vertical datum. The Cape Cod Bay tide record was used as the open boundary condition of the hydrodynamic model. Data from inside the system were used to calibrate the model. In July, the Bridge Creek gauge (B5) was discovered to have been moved at the end of June, 19 days after its initial deployment. The gauge was redeployed on July 22.

Tide records longer than 29 days are necessary for a complete evaluation of tidal dynamics within the estuarine system. Although a one-month record likely does not include extreme high or low tides, it does provide an accurate basis for typical tidal conditions governed by both lunar and solar motion. For numerical modeling of hydrodynamics, the typical tide conditions associated with a one-month record are appropriate for driving tidal flows within the estuarine system.

Plots of the tide data from the five gauges are shown in Figure V-3 for a 48-day period during the gauge deployment. The spring-to-neap variation in tide range is easily recognizable in these plots. The data record begins during a transitional period from spring to neap tides. The first quarter moon occurs June 24, during the first period of diminished neap tides. After this, there is a period of spring tides around July 3, which occurs around the time of the new moon of July 1. The minimum neap tide range in the offshore record is 2.0 feet (July 24), while the maximum spring tide rage (occurring about a week earlier) is 13.4 feet (August 1).

A visual comparison between tide elevations offshore and at the different stations in the system shows that the tide amplitude in the upper reaches is controlled by the bottom elevation of these areas (Figure V-4). There is only a minor reduction of the water elevation at times of high tide (about 5 inches) between the offshore and inshore areas. Low tide elevations are highest at the Bridge Creek gauge, where the marsh creek channel bottom controls the minimum water levels. At Bridge Creek the low tide elevation during spring tides can be nearly 4 feet higher than offshore in Cape Cod Bay.

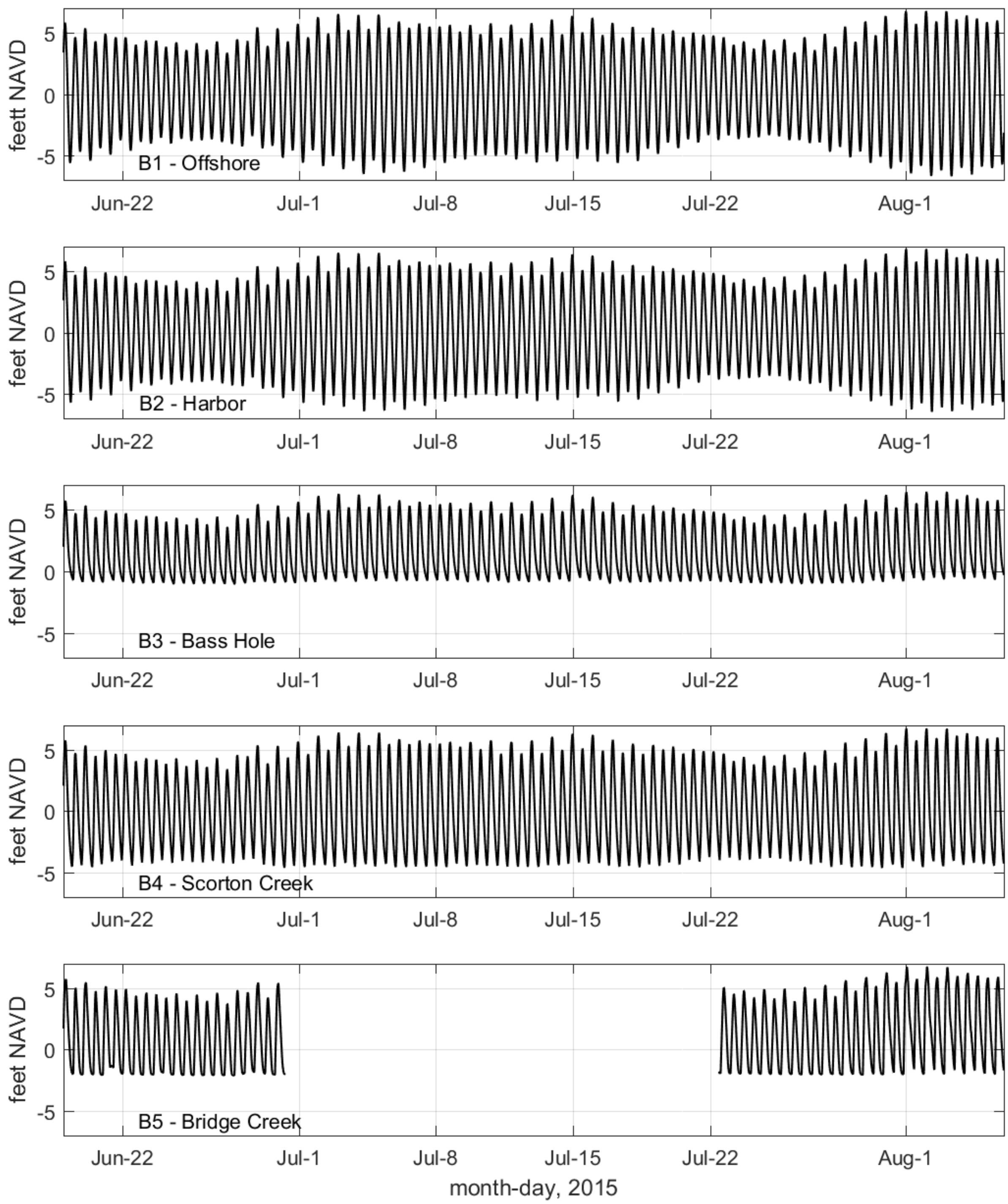


Figure V-3. Plots of observed tides for stations in Barnstable Harbor, for the 48-day period between June 19 and August 6, 2015. All water levels are referenced to the NAVD vertical datum.

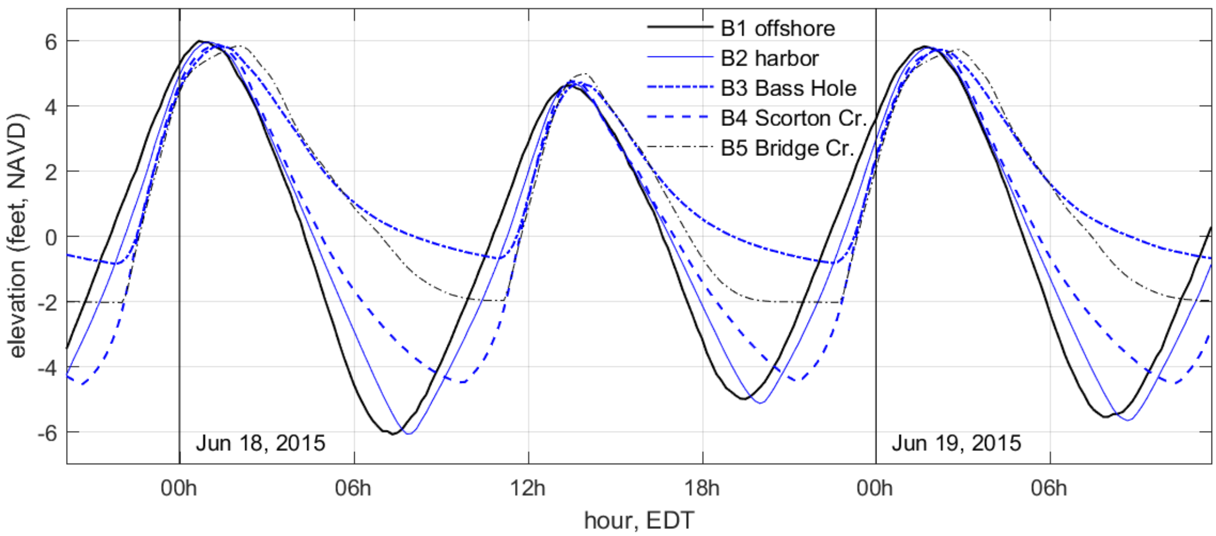


Figure V-4. Four-day tide plot showing tides measured in Cape Cod Bay and at stations in the Barnstable Great Marsh estuary system.

V.2.2.1 Tide Datums

To better quantify the changes to the tide from the inlet to inside the system, the standard tide datums were computed from a 30-day subset of the tide records. These datums are presented in Table V-1. For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data was available; however, these datums still provide a useful comparison of tidal dynamics within the system. The Mean Higher High (MHH) and Mean Lower Low (MLL) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW.

Tidal damping in the upper reaches of the Harbor system is seen by the elevated levels of the low water datums, as presented in Table V-1. For example, MLW at the Bridge Creek station is more than four feet higher than it is in the main basin of the Harbor. Though the tide range is truncated in in the marsh channels, the tidal flows are conveyed across the marsh very efficiently, as can be seen by the small amount of different in elevation of high tide at all gauging stations. This is the case even at the Bridge Creek gauging station which is nearly two miles from the open water of the Harbor, where MHW is the same as it is offshore.

V.2.2.2 Tide Harmonic Analysis

A more thorough harmonic analysis of the tidal time series was also performed to produce tidal amplitude and phase of the major tidal constituents, and provide assessments of hydrodynamic ‘efficiency’ of the system in terms of tidal attenuation. This analysis also yielded an assessment of the relative influence of non-tidal, or residual, processes (such as wind forcing) on the hydrodynamic characteristics of each system.

A harmonic analysis was performed on the time series from each gauge location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The observed astronomical tide is the sum of several individual tidal constituents, with a particular amplitude and frequency. For demonstration purposes a graphical example of how these constituents add together is shown in Figure V-5. In this case, the amplitudes and phase of 21 known tidal constituents were computed. Table V-2 presents the amplitudes of eight tidal constituents computed for the Barnstable Harbor station records. The M_2, or the familiar twice-a-day lunar semi-diurnal tide, is the strongest contributor to the signal with an offshore amplitude of 4.7 feet. The total range of the M_2 tide is twice the amplitude, or 10.4 feet.

Table V-1. Tide datums computed from 50-day records collected offshore and in the Barnstable Great Marsh estuary system in June, July and August 2015. Datum elevations are given relative to NAVD vertical datum.

Tide Datum	Cape Cod Bay	Harbor	Bass Hole	Scorton Creek	Bridge Cr.
Maximum Tide	6.7	6.8	6.7	6.7	6.5
MHHW	5.5	5.5	5.4	5.4	5.4
MHW	5.1	5.1	5.1	5.1	5.1
MTL	0.1	0.1	1.6	0.4	2.2
MLW	-4.9	-4.9	-1.9	-4.3	-0.7
MLLW	-5.1	-5.2	-2.0	-4.4	-0.9
Minimum Tide	-6.7	-6.4	-2.1	-4.6	-1.0
Mean Range	9.9	10.0	7.0	9.4	5.8

The diurnal constituents (once daily), K_1 and O_1, have amplitudes of approximately 0.5 feet and 0.3 respectively. Other semi-diurnal tides, the S_2 (12.00 hour period), N_2 (12.66-hour period) and L_2 (12.19-hour period) tides, also contribute to the total tide signal, with amplitudes of 0.6 feet, 0.9 feet and 0.2 feet, respectively.

The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 period for the M_6) and result from frictional attenuation of the M_2 tide in shallow water. While the main diurnal and semi-diurnal constituents tend to decrease in the system, the M_4 is seen to increase by 0.7 feet between Cape Cod Bay and Scorton Creek as energy is transferred from the M_2 to its harmonics. Generally, it can be seen that as the total tide range is attenuated through the system there is a corresponding reduction in the amplitude of the individual tide constituents.

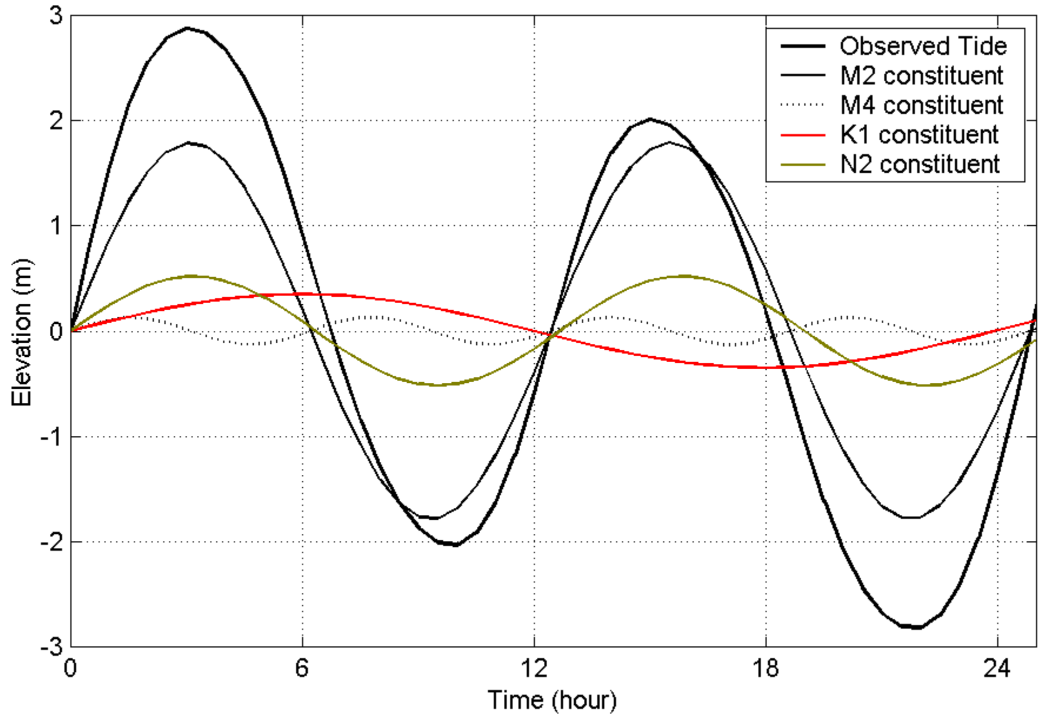


Figure V-5. Example of an observed astronomical tide as the sum of its primary constituents.

Table V-2. Tidal Constituents computed for tide stations in the Barnstable Great Marsh estuary system and offshore in Cape Cod Bay, June, July and August 2015. Constituents for Bridge Creek rely on data from August September 2015, due to the gauge failure in July

	Amplitude (feet)							
Constituent	M_2	M_4	M_6	K_1	N_2	S_2	O_1	L_2
Period (hours)	12.42	6.21	4.14	23.93	12.66	12.00	25.82	12.19
Cape Cod Bay	4.72	0.10	0.18	0.51	0.89	0.55	0.32	0.15
Harbor	4.59	0.29	0.24	0.49	0.83	0.51	0.32	0.16
Bass Hole	2.66	0.84	0.24	0.36	0.39	0.25	0.34	0.17
Scorton Creek	4.31	0.86	0.11	0.44	0.69	0.41	0.34	0.25
Bridge Creek	3.36	0.72	0.21	0.33	0.62	0.36	0.41	0.45

Together with the change in constituent amplitude across Harbor, the phase change of the tide is easily seen from the results of the harmonic analysis. Table V-3 shows the delay of the M_2 at different points in the Barnstable Great Marsh estuary system, relative to the timing of the M_2 constituent in Cape Cod Bay, at the harbor entrance. The largest delay occurs at the Bridge Creek station, where the M_2 phase is offset by one hour.

In addition to the tidal analysis, the data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. These other processes include wind forcing (set-up or set-down) within the estuary, as well as sub-tidal oscillations of the sea surface. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large compared to tidal flow.

The results of an analysis to determine the energy distribution (or variance) of the measured water elevation records for the gauge records in Barnstable Harbor compared to the energy content of the astronomical tidal signal (re-created by summing the contributions from the 21 constituents determined by the harmonic analysis) is presented in Table V-4. Subtracting the tidal signal from the original elevation time series resulted in the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. Figure V-6 shows the comparison of the measured tide from Cape Cod Bay, with the computed astronomical tide resulting from the harmonic analysis, and the resulting non-tidal residual.

Table V-4 shows that the variance of tidal energy was largest in the offshore signal, as should be expected. The analysis also shows that tides are responsible for nearly 100% of the water level changes in Cape Cod Bay and all of Barnstable Harbor for the gauge deployment period. This indicates that the hydrodynamics of the system is influenced predominantly by astronomical tides. The non-tidal residual is largest by percentage in the Bridge Creek, where the non-tidal component is 5% of the total variance of the observed water level changes.

Table V-3. M_2 tidal constituent phase delay (relative to the Cape Cod Bay station) for gauge locations in the Barnstable Great Marsh estuary system, determined from measured tide data.

Station	Delay (minutes)
Harbor	18.0
Bass Hole	47.1
Scorton Creek	45.6
Bridge Creek	58.0

Table V-4. Percentages of Tidal versus Non-Tidal Energy for stations in the Barnstable Great Marsh estuary system and Cape Cod Bay, June, July and August, 2015.

TDR Location	Total Variance (ft^2)	Tidal (%)	Non-tidal (%)
Cape Cod Bay	11.7	100.0	0.0
Harbor	11.1	99.5	0.5
Bass Hole	4.2	96.6	3.4
Scorton Creek	10.1	99.0	1.0
Bridge Creek	6.6	94.7	5.3

V.2.2.3 Tide Flood and Ebb Dominance

An investigation of the flood or ebb dominance of different areas in the Barnstable Great Marsh estuary system was performed using the measured tide data. Estuaries and sub-embayments that are flood dominant are typically areas that collect sediment over time since they have maximum flood tide velocities that are greater than the maximum velocities that occur during the ebb portion of the tide. Salt marshes tend to be flood dominant, as this condition allows them to collect material that is required to maintain healthy marsh resources.

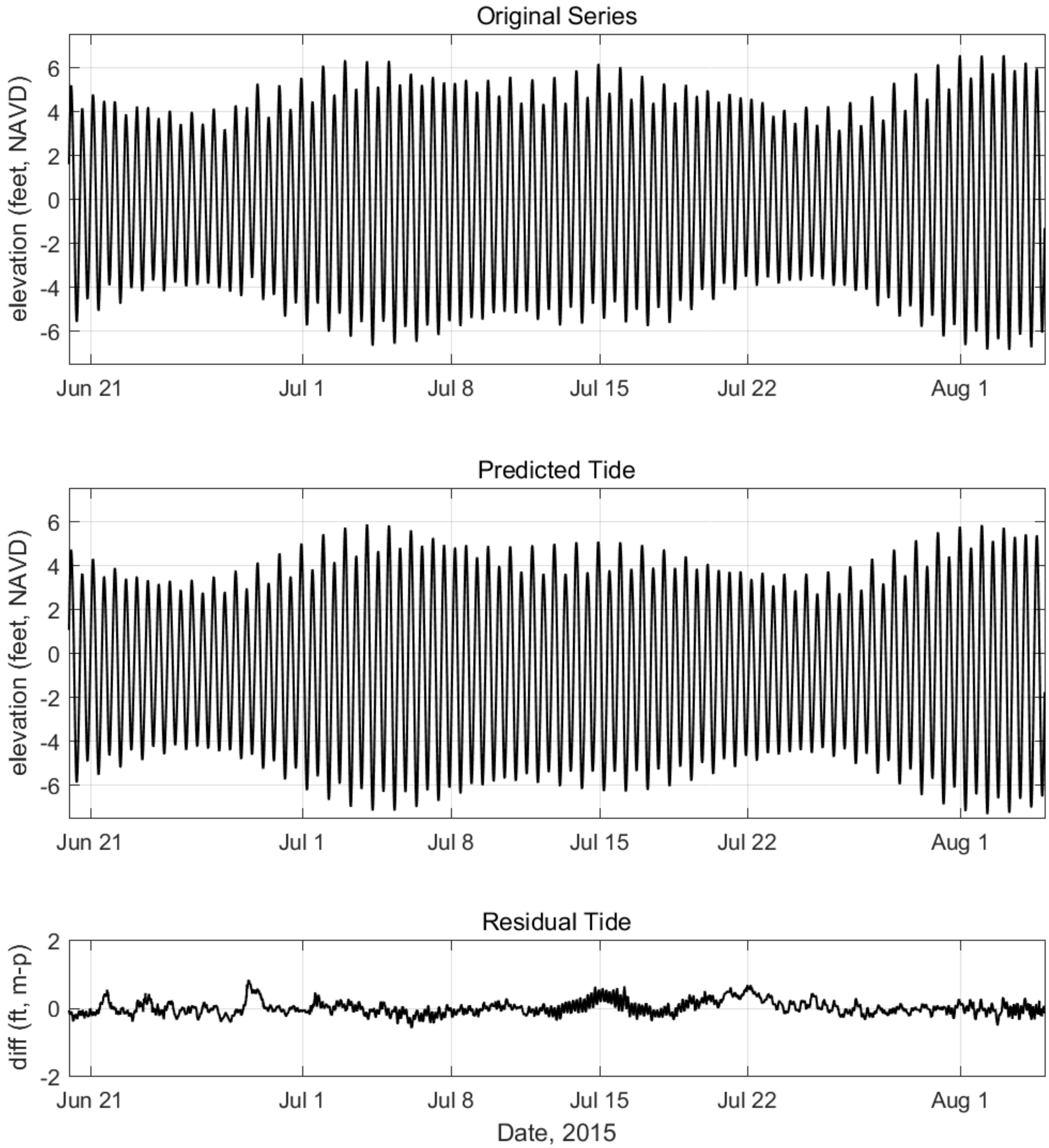


Figure V-6. Plot showing the comparison between the measured tide time series (top plot), and the predicted astronomical tide (middle plot) computed using the 21 individual tide constituents determine in the harmonic analysis of the Cape Cod Bay gauge data, collected offshore Barnstable Harbor. The residual tide shown in the bottom plot is computed as the difference between the measured and predicted time series (r=m-p).

Flood or ebb dominance in channels of a tidal system can be determined by utilizing the results of the harmonic analysis of tidal elevations, or by performing a similar analysis on a time series of tidal currents. A discussion of the method of relative phase determination is presented in Friedrichs and Aubrey (1988). For this method, the same M_2 and M_4 tidal constituents presented in Table V-2 were used as the basis of this analysis.

For constituents based on tidal elevations, the relative phase difference is computed as the difference between two times, the M_2 phase and the phase of the M_4, expressed as $\Phi=2M_2-M_4$. If Φ is between 0 and 180 degrees ($0<\Phi<180$), then the channel is characterized as being flood dominant, and peak flood velocities will be greater than for peak ebb. Alternately, if Φ were between 180 and 360 degrees ($180<\Phi<360$), then the channel would be ebb dominant. If Φ is exactly 0 or 180 degrees, neither flood nor ebb dominance occurs. For Φ equal to exactly 90 or 270 degrees, maximum tidal distortion occurs and the velocity residuals of a channel are greatest. This relative phase relationship is presented graphically in Figure V-7.

Though this method of tidal constituent analysis provides similar results to a visual inspection of a tidal record (e.g., by comparing peak ebb and flood velocities), it allows a more exact characterization of the tidal processes. By this analysis technique, a channel can be characterized as being strongly, moderately, or weakly flood or ebb dominant.

The five gauge stations in the harbor were used for this analysis. These data make it possible to characterize the flood or ebb dominance of different areas of the system from offshore (W-1 in Cape Cod Bay) through to the upper reaches of the main tidal creeks (e.g., W-4, upstream of Uncle Tim's Bridge). The results of this velocity analysis of the Barnstable Harbor measured tide data show that although the offshore gauge is ebb dominant, all interior gauge stations indicate flood dominance. The computed values of $2M_2-M_4$ are presented in Table V-5.

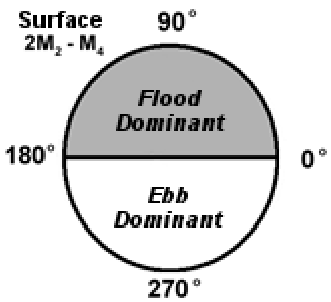


Figure V-7. Relative velocity phase relationship of M2 and M4 tidal elevation constituents and characteristic dominance, indicated on the unit circle. Relative phase is computed as the difference of two times the M2 phase and the M4 phase (2M2-M4). A relative phase of exactly 90 or 270 degrees indicates a symmetric tide, which is neither flood nor ebb dominant.

Table V-5. Barnstable Harbor relative tidal phase differences of M_2 and M_4 tide constituents, determined using tide elevation record records.

location	$2M_2-M_4$ relative phase (deg)	Characteristic dominance
Cape Cod Bay, B1	209.5	moderate ebb
Harbor basin, B2	86.4	strong flood
Bass Hole, B3	49.3	moderate flood
Scorton Creek, B4	69.1	moderate flood
Bridge Creek, B5	52.0	moderate flood

V.3 HYDRODYNAMIC MODELING

For the modeling of the Barnstable Great Marsh estuary system, MEP Technical Team members from Applied Coastal Research and Engineering (ACRE) utilized a hydrodynamic computer model to evaluate tidal circulation and flushing in the Harbor. The particular model employed was the RMA-2 model developed by Resource Management Associates (King, 1990). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers. Applied Coastal staff members have utilized RMA-2 for numerous flushing studies on Cape Cod under the MEP umbrella, including Sandwich Harbor, Wellfleet Harbor, Popponesset Bay, Nantucket Harbor, Falmouth "finger" Ponds (Howes *et al*, 2005), Three Bays (Kelley *et al*, 2003) and Barnstable Harbor (Wood, *et al*, 1999).

V.3.1 Model Theory

In its original form, RMA-2 was developed by William Norton and Ian King under contract with the U.S. Army Corps of Engineers (Norton et al., 1973). Further development included the introduction of one-dimensional elements, state-of-the-art pre- and post-processing data programs, and the use of elements with curved borders. Recently, the graphic pre- and post-processing routines were updated by Brigham Young University through a package called the Surfacewater Modeling System or SMS (BYU, 1998). Graphics generated in support of this report primarily were generated within the SMS modeling package.

RMA-2 is a finite element model designed for simulating one- and two-dimensional depth-averaged hydrodynamic systems. The dependent variables are velocity and water depth, and the equations solved are the depth-averaged Navier Stokes equations. Reynolds assumptions are incorporated as an eddy viscosity effect to represent turbulent energy losses. Other terms in the governing equations permit friction losses (approximated either by a Chezy or Manning formulation) for Coriolis effects and surface wind stresses. All the coefficients associated with these terms may vary from element to element. The model utilizes quadrilaterals and triangles to represent the prototypical system. Element boundaries may either be curved or straight.

The time dependence of the governing equations is incorporated within the solution technique needed to solve the set of simultaneous equations. This technique is implicit; therefore, unconditionally stable. Once the equations are solved, corrections to the initial estimate of velocity and water elevation are employed, and the equations are re-solved until the convergence criteria is met.

V.3.2 Model Setup

There are three main steps required to implement RMA-2:

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using 2009 and 2014 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of the Barnstable Harbor grid based on the tide gauge data collected offshore in Cape Cod Bay. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several model calibration simulations for the system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.3.2.1 Grid generation

The grid generation process was aided by the use of the SMS package. Digital aerial orthophotos, the summer 2015 bathymetry survey data, and available 2013 LiDAR topography were imported to SMS, and a finite element grid was generated to represent the estuary. The aerial photograph was used to determine the land boundary of the system, as well as determine the surface coverage of salt marsh. The bathymetry and topography data were interpolated to the developed finite element mesh of the system. The completed grid consists of 19,920 nodes, which describe 7,885 total 2-dimensional (depth averaged) quadratic elements. The maximum nodal depth is -36ft (NGVD) in the natural channel of the harbor. The completed grid mesh of the Barnstable Great Marsh estuary system is shown in Figure V-8.

The finite element grid for the system provides the detail necessary to accurately evaluate the variation in hydrodynamic properties of Barnstable Harbor. Areas of marsh were included in the model. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary. Grid resolution is generally governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability of the system. Relatively fine grid resolution is employed where complex flow patterns are expected, generally near the inlet. Appropriate implementation of wider node spacing and larger elements reduces computer run time with no sacrifice of accuracy.

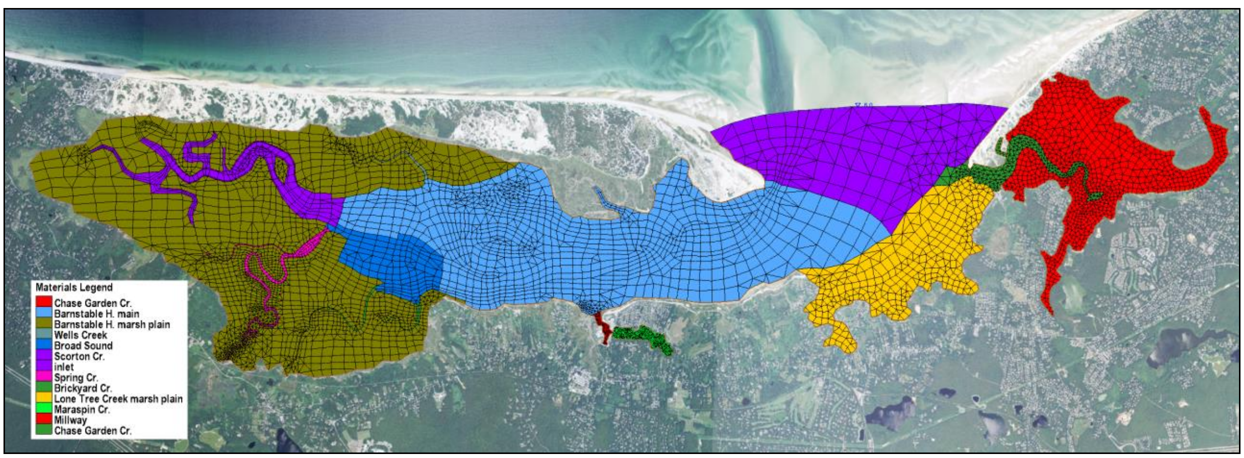


Figure V-8. Plot of hydrodynamic model grid mesh for Barnstable Harbor. Colors are used to designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.3.2.2 Boundary condition specification

Three types of boundary conditions were employed for the RMA-2 model of the Barnstable Great Marsh estuary system: 1) "slip" boundaries, 2) tidal elevation boundaries, and 3) constant flow input boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. A tidal boundary condition was specified using the data collected at the offshore gauge station. TDR measurements provided the required data. The rise and fall of the tide in the Bay is the primary driving force for estuarine circulation in this system. Dynamic (time-varying) model simulations specified a new water surface elevation at the open boundary of the Barnstable Harbor grid every model time step. The model runs of the Harbor used a 10-minute time step, which the same as the 10-minute sampling rate of the measured tide data. Details concerning the constant flow input boundary conditions included in the hydro model are discussed in Section VI.

V.3.2.3 Calibration

After developing the finite element grids, and specifying boundary conditions, the model for the Barnstable Great Marsh estuary system was calibrated. The calibration procedure ensures that the model accurately predicts what was observed in nature during the field measurement program. Numerous model simulations are typically required for an estuary model, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

Calibration of the hydrodynamic model required a close match between the modeled and measured tides from stations inside the system (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured tides.

Once visual agreement was achieved, a 14.5-day period (28 tide cycles) was modeled to calibrate the model based on dominant tidal constituents discussed in Section V.2. The half-lunar-month period was extracted from a longer simulation to avoid effects of model spin-up, and to focus on average tidal conditions. Modeled tides for the calibration time period were evaluated for time (phase) lag and height damping of dominant tidal constituents. The calibration was performed for the 14.5-day period beginning July 22, 2015 at 0430 EDT.

After the model was calibrated, an additional model run was made in order corroborate the model performance in a time period outside of the calibration period. The model corroboration run period is 10-days long and begins June 20, 2015 at 0300 EDT.

The completed model was used to analyze existing detailed flow patterns and compute residence times. The flushing analysis used the model calibration period. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. For instance, average residence times were computed over the entire seven-day simulation. Other methods, such as dye and salinity studies, evaluate tidal flushing over relatively short time periods (less than one day). These short-term measurement techniques may not be representative of average conditions due to the influence of unique, short-lived atmospheric events.

V.3.2.3.a Friction coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude damping and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient, and is applied to grid areas by user specified material types. Initially, a Manning's friction coefficient value of 0.020 was specified for all element material types. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels and marsh plains with higher friction (Henderson, 1966).

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning's coefficients were matched to bottom type. For example, lower friction coefficients were specified for the main marsh creeks, versus the extensive marsh plain areas of the Harbor, which provide greater flow resistance by the presence of marsh vegetation. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were initially selected based on ranges provided by the available engineering references (Chow, 1959). Values were incrementally changed as appropriate to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-6.

Table V-6. Manning's Roughness and eddy viscosity coefficients used in simulations of the Barnstable Great Marsh estuary system. These embayment delineations correspond to the material type areas shown in Figure V-9.

System Embayment	bottom friction	eddy viscosity lb-sec/ft^2
Harbor Inlet	0.02	200
Great Marshes creeks	0.03	80
Great Marshes marsh plain	0.05	80
Millway	0.02	80
Chase Garden Creek	0.03	80
Chase Garden Greek marsh plain	0.07	80
Lone Tree marsh creek plain	0.07	100
Maraspin creek marsh plain	0.07	100

V.3.2.3.b Turbulent exchange coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). In most cases, the modeled systems were relatively insensitive to turbulent exchange coefficients because there were no regions of strong turbulent flow. Typically, model turbulence coefficients were set between 80 and 100 lb-sec/ft^2 (Table V-6). A higher value was used in the inlet region.

V.3.2.3.c Marsh porosity processes

Modeled hydrodynamics were complicated by wetting/drying cycles on the marsh plain included in the model of the Barnstable Harbor/Chase Garden Creek system. Cyclically wet/dry areas of the marsh will tend to store waters as the tide begins to ebb and then slowly release water as the water level drops within the creeks and channels. This store-and-release characteristic of these marsh regions was partially responsible for the distortion of the tidal signal and the elongation of the ebb phase of the tide. On the flood phase, water rises within the channels and creeks initially until water surface elevation reaches the marsh plain, when at this point the water level remains nearly constant as water 'fans' out over the marsh surface. The rapid flooding of the marsh surface corresponds to a flattening out of the tide curve approaching high water. Marsh porosity is a feature of the RMA-2 model that permits the modeling of hydrodynamics in marshes and tide flats. This model feature essentially simulates the store-and-release capability of the marsh plain by allowing grid elements to transition gradually between wet and dry states. This technique allows RMA-2 to change the ability of an element to hold water, similar to a sponge.

V.3.2.3.d Comparison of modeled tides and measured tide data

A best-fit of model output for the measured data was achieved using the aforementioned values for friction and turbulent exchange. Figures V-9 through V-13 illustrate sections of the 8-day simulation periods for the calibration model. Modeled (solid line) and measured (dotted line) tides are illustrated at each model location with a corresponding TDR.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration was required to quantify the accuracy of the models. Calibration of the M_2 harmonic was the highest priority since M_2 accounted for a majority of the forcing tide energy throughout the system. Four tidal constituents were selected for constituent comparison: the K_1, M_2, M_4 and M_6. After calibrating the model, its performance was further corroborated by running the model for an additional verification time period (August 17 through August 24, 2008).

Measured tidal constituent amplitudes are shown in Table V-7 for the calibration and Table V-9 for the verification simulation. The constituent amplitudes shown in this table differ from those in Table V-2 because constituents were computed for only the separate 7-day sub-sections of the month-long period represented in Table V-2. In Tables V-8 and V-10, error statistics are shown for the calibration and verification.

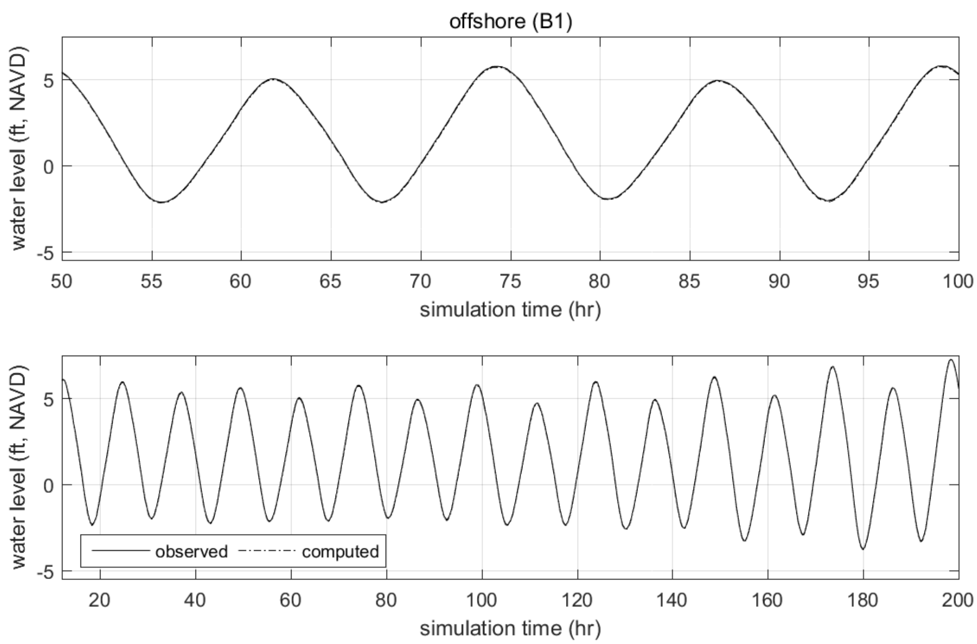


Figure V-9. Comparison of model output and measured tides for the offshore TDR station for the calibration model run (July 21, 2015 at 1600 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

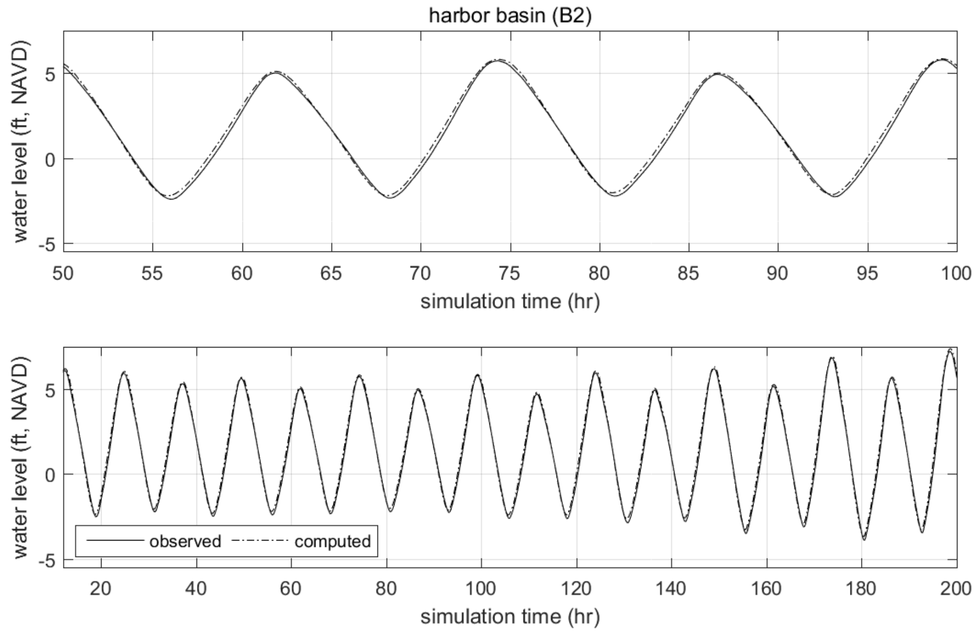


Figure V-10. Comparison of model output and measured tides for the TDR at the main harbor basin station (B2) for the calibration model run (July 21, 2015 at 1600 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

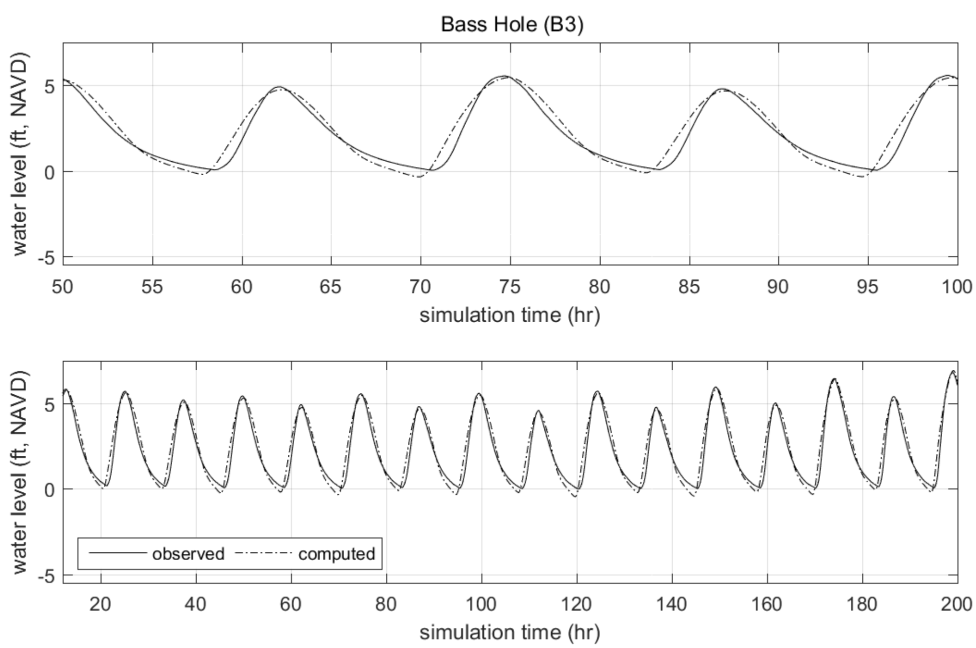


Figure V-11. Comparison of model output and measured tides for the Bass Hole TDR location (B3) for the final calibration model run (July 21, 2015 at 1600 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

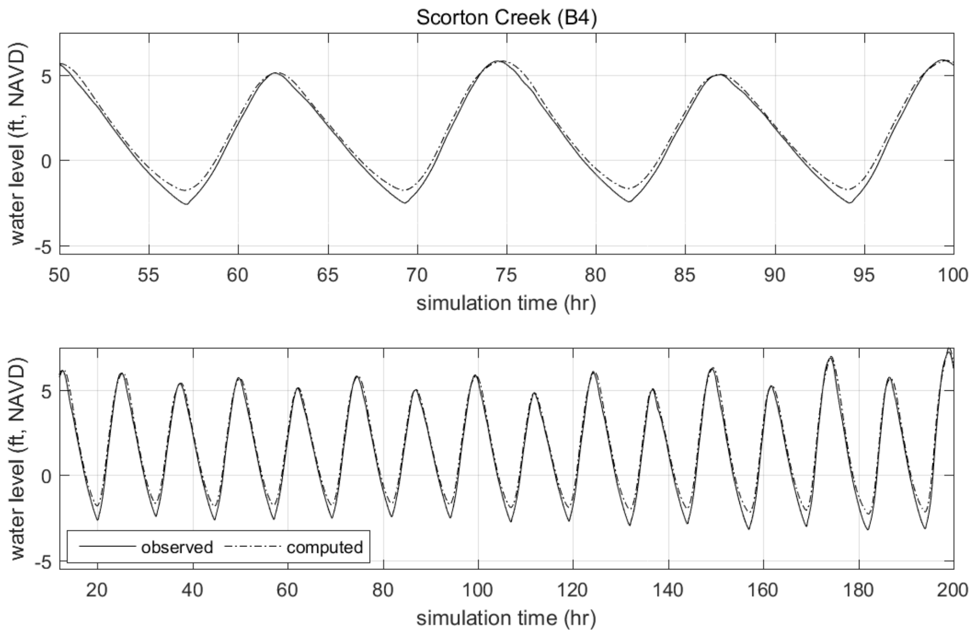


Figure V-12. Comparison of model output and measured tides for the Scorton Creek TDR station (B4) for the final calibration model run (July 21, 2015 at 1600 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

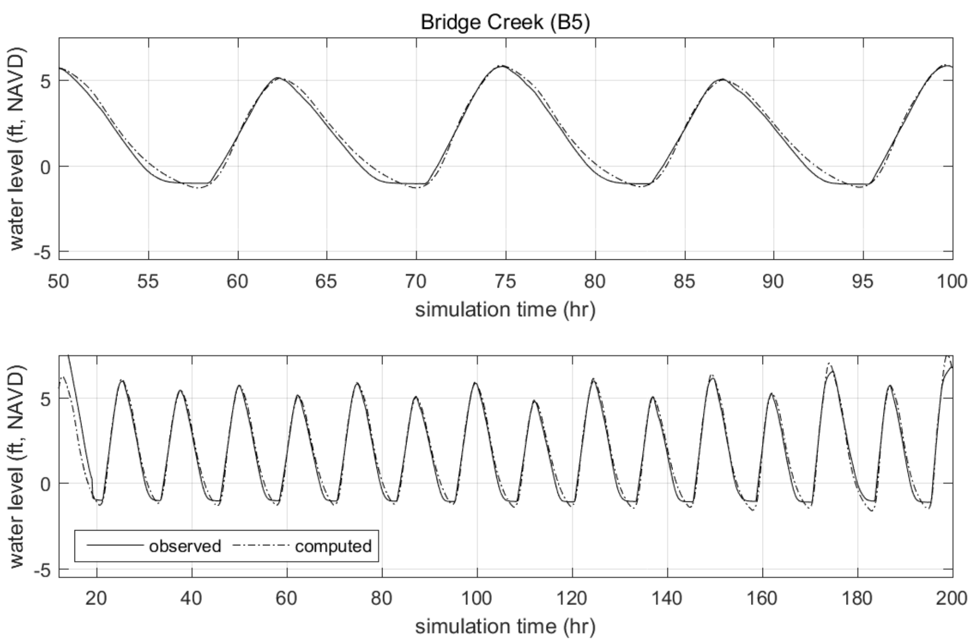


Figure V-13. Comparison of model output and measured tides for the Bridge Creek TDR station (B5) for the final calibration model run (July 21, 2015 at 1600 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

The constituent calibration resulted in excellent agreement between modeled and measured tides. The errors associated with tidal constituent amplitude for both the calibration and verification simulations were on the order of 0.1 ft, which is of the same order of magnitude accuracy as that of the tide gauges (0.25 ft). Time lag errors for the main estuary reach were generally less than the time increment resolved by the model and tide data (10 minutes), indicating good agreement between the model and data. The skill of the model calibration is also demonstrated by the high degree of correlation (R^2) and low RMS error shown in Table V-9 for all stations.

Table V-7. Tidal constituents for measured water level data and calibrated model output, with model error amplitudes, for Barnstable Harbor, during modeled calibration time period.

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕK_1
Cape Cod Bay	4.16	0.10	0.18	0.58	19.2	52.4
Harbor	4.13	0.09	0.20	0.57	25.0	57.3
Bass Hole	2.73	0.44	0.13	0.51	42.5	94.2
Scorton Creek	3.79	0.53	0.07	0.54	40.5	72.8
Bridge Creek	3.52	0.70	0.18	0.54	52.1	80.0
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕK_1
Cape Cod Bay	4.16	0.09	0.18	0.58	19.8	52.6
Harbor	4.09	0.22	0.23	0.55	28.4	59.0
Bass Hole	2.49	0.81	0.22	0.45	46.7	91.9
Scorton Creek	4.00	0.68	0.02	0.51	41.8	71.8
Bridge Creek	3.36	0.62	0.18	0.50	47.8	103.8
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕK_1
Cape Cod Bay	0.00	0.01	0.00	0.00	-1.2	-0.8
Harbor	0.04	-0.13	-0.03	0.02	-7.0	-6.8
Bass Hole	0.24	-0.37	-0.09	0.06	-8.7	9.2
Scorton Creek	-0.21	-0.15	0.05	0.03	-2.7	4.0
Bridge Creek	0.16	0.08	0.00	0.04	8.9	-94.9

Table V-8. Error statistics for the Barnstable Harbor hydrodynamic model, for model calibration.

	R^2	RMS error (feet)
Cape Cod Bay	1.00	0.0
Harbor	0.99	0.2
Bass Hole	0.94	0.5
Scorton Creek	0.98	0.5
Bridge Creek	0.97	0.4

V.3.2.4 Hydrodynamic Model Corroboration

An additional corroboration model run was made to verify the performance of the hydrodynamic model over a span of time that does not overlap the calibration time period. The 10-day period between 0300 June 20 and 0300 June 30, 2015 was selected for the corroboration run since this was an available period when data were recovered from all five tide gauges.

Table V-9. Tidal constituents for measured water level data and calibrated model output, with model error amplitudes, for Barnstable Harbor, during modeled corroboration time period.

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕK_1
Cape Cod Bay	4.00	0.08	0.18	0.41	-15.9	-65.7
Harbor	4.00	0.08	0.20	0.40	-10.1	-60.1
Bass Hole	2.69	0.43	0.15	0.30	7.7	-39.0
Scorton Creek	3.71	0.52	0.05	0.37	5.7	-46.6
Bridge Creek	3.45	0.67	0.17	0.36	18.0	-39.8
Measured tide during corroboration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕK_1
Cape Cod Bay	4.00	0.09	0.18	0.41	-15.2	-65.2
Harbor	3.95	0.19	0.25	0.39	-6.3	-55.8
Bass Hole	2.41	0.82	0.24	0.27	12.5	-40.7
Scorton Creek	3.86	0.68	0.04	0.34	7.6	-41.6
Bridge Creek	3.26	0.60	0.21	0.29	12.0	-31.1
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕK_1
Cape Cod Bay	0.00	0.00	0.00	0.00	-1.4	-2.1
Harbor	0.05	-0.11	-0.05	0.01	-7.8	-17.2
Bass Hole	0.28	-0.39	-0.09	0.03	-9.8	6.6
Scorton Creek	-0.15	-0.16	0.02	0.03	-3.9	-19.8
Bridge Creek	0.19	0.07	-0.04	0.07	12.4	-34.7

Table V-10. Error statistics for the Barnstable Harbor hydrodynamic model, for model corroboration.

	R^2	RMS error (feet)
Cape Cod Bay	1.00	0.01
Harbor	0.99	0.20
Bass Hole	0.91	0.55
Scorton Creek	0.96	0.54
Bridge Creek	0.97	0.43

V.3.3 Model Circulation Characteristics

The final calibrated model serves as a useful tool in investigating circulation characteristics of the whole Barnstable Great Marsh estuary system. Inputs of bathymetry and tide data can be leveraged to develop further insight into tidal velocities and flow rates at any point in the model domain. This is a very useful feature of a hydrodynamic model, where a limited amount of collected data can be expanded to determine the physical attributes of the system in areas where no physical data record exists. As an example, Figure V-14 shows color contours and vectors that indicate velocity during a single model time step, during a period of maximum flood currents at the inlet.

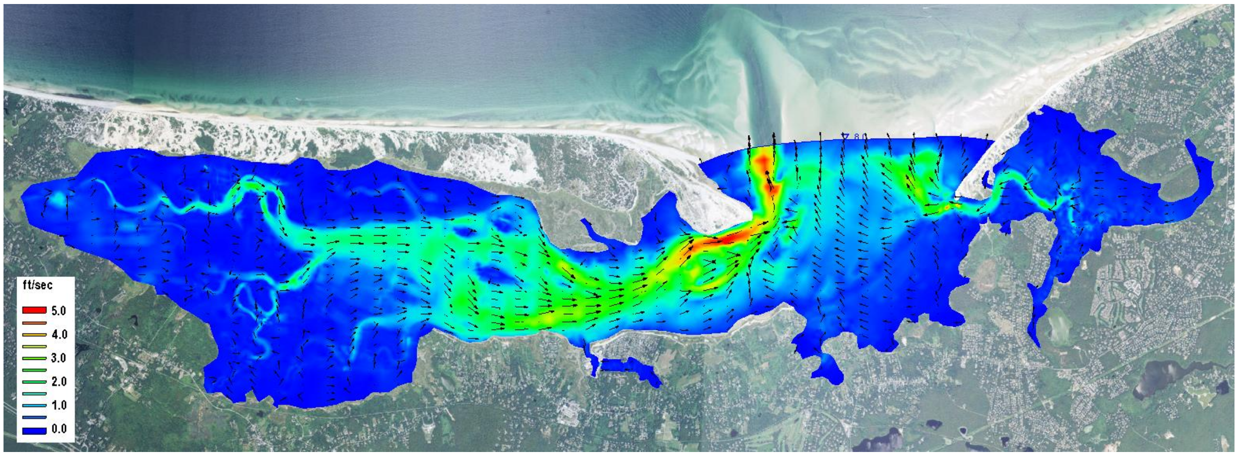


Figure V-14. Example of Barnstable Harbor hydrodynamic model output for a single time step during an ebbing tide. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

As another example, from the calibration model run of the Barnstable Great Marsh estuary system, the total flow rate of water flowing through the inlet culvert can be computed with the hydrodynamic model. The variation of flow as the tide floods and ebbs is seen in the plot of system flow rates in Figure V-15. During spring tides, the maximum flood flow rates into the harbor reach 191,500 ft^3/sec. Maximum ebb flow rates during spring tides are less than half of the flow rates experienced during spring tides (90,000 ft^3/sec).

V.3.4 Flushing Characteristics

Since the magnitude of freshwater inflow is much smaller in comparison to the tidal exchange through the inlet, the primary mechanism controlling estuarine water quality within the modeled Barnstable Great Marsh estuary system is tidal exchange. A rising tide offshore in Cape Cod Bay creates a slope in water surface from the ocean into the upper-most reaches of the modeled system. Consequently, water flows into (floods) the system. Similarly, the estuary drains into the open waters of Cape Cod Bay on an ebbing tide. This exchange of water between the system and the ocean is defined as tidal flushing. The calibrated hydrodynamic model is a tool to quantitatively evaluate tidal flushing of the harbor system, and was used to compute flushing rates (residence times) and tidal circulation patterns.

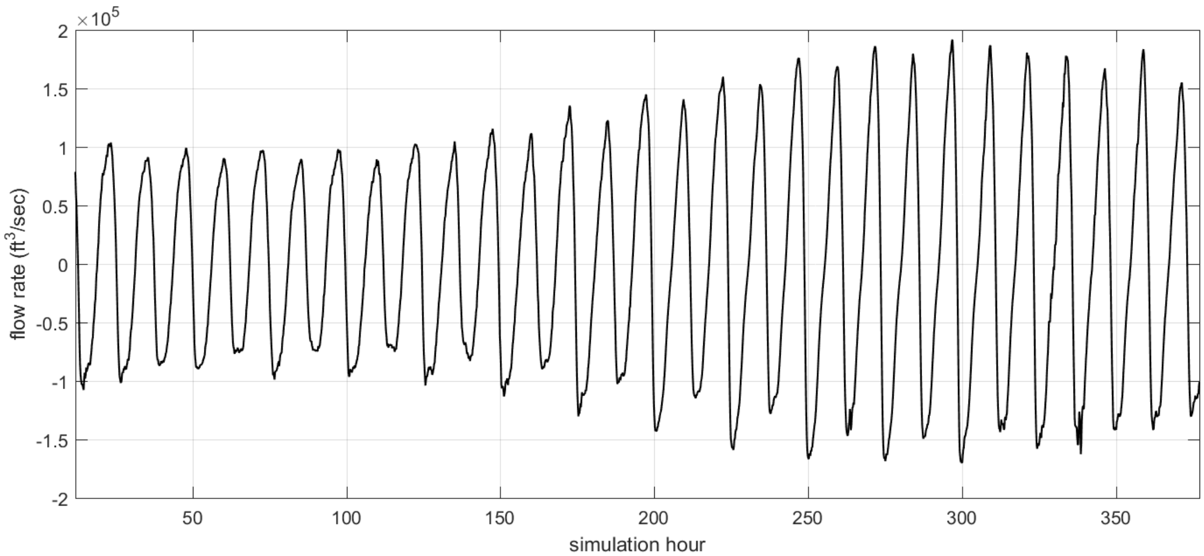


Figure V-15. Time variation of computed flow rates for the whole of the Barnstable Great Marsh estuary system. Model period shown corresponds to spring tide conditions, where the tide range is the largest, and resulting flow rates are correspondingly large compared to neap tide conditions. Positive flow indicated flooding tide flows, while negative flow indicates ebbing tide flows.

Flushing rate, or residence time, is defined as the average time required for a parcel of water to migrate out of an estuary from points within the system. For this study, system residence times were computed as the average time required for a water parcel to migrate from a point within the embayment to the entrance of the system. System residence times are computed as follows:

$$T_{system} = \frac{V_{system}}{P} t_{cycle}$$

where T_{system} denotes the residence time for the system, V_{system} represents volume of the (entire) system at mean tide level, P equals the tidal prism (or volume entering the system through a single tidal cycle), and t_{cycle} the period of the tidal cycle, typically 12.42 hours (or 0.52 days). To compute system residence time for a sub-embayment, the tidal prism of the sub-embayment replaces the total system tidal prism value in the above equation.

In addition to system residence times, a second residence, the local residence time, was defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. Using the Millway as an example, the system residence time is the average time required for water to migrate out of the Millway and Maraspin Creek, then across the main basin of Barnstable Harbor, and finally out through the harbor inlet and into Cape Cod Bay. Alternatively, the local residence time is the average time required for water to migrate from the inner harbor and into the main basin of the Harbor (not all the way to the Bay). Local residence times for each sub-embayment are computed as:

$$T_{local} = \frac{V_{local}}{P} t_{cycle}$$

where T_{local} denotes the residence time for the local sub-embayment, V_{local} represents the volume of the sub-embayment at mean tide level, P equals the tidal prism (or volume entering the local sub-embayment through a single tidal cycle), and t_{cycle} the period of the tidal cycle (again, 0.52 days).

Residence times are provided as a first order evaluation of estuarine water quality. Lower residence times generally correspond to higher water quality; however, residence times may be misleading depending upon pollutant/nutrient loading rates and the overall quality of the receiving waters. As a qualitative guide, system residence times are applicable for systems where the water quality within the entire estuary is degraded and higher quality waters provide the only means of reducing the high nutrient levels.

The rate of pollutant/nutrient loading and the quality of water outside the estuary both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. It is impossible to evaluate an estuary's health based solely on flushing rates. Efficient tidal flushing (low residence time) is not an indication of high water quality if pollutants and nutrients are loaded into the estuary faster than the tidal circulation can flush the system. Neither are low residence times an indicator of high water quality if the water flushed into the estuary is of poor quality. Advanced understanding of water quality is obtained from applying the calibrated hydrodynamic model as described in the following section of this report (Section VI) and by extending the model to include pollutant/nutrient dispersion. The water quality model provides an additional valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Harbor system.

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in the system, model results were used to compute residence times. Residence times were computed for the entire estuary, as well as two subdivisions of the system. In addition, system and local residence times were computed to indicate the range of conditions possible for the system.

Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged over a flood tidal cycle (tidal prism). The mean volumes and tide prisms of the four system divisions used in this analysis are presented in Table V-11.

Table V-11. Barnstable Harbor mean volume and average tidal prism during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Barnstable Harbor/Chase Garden Cr. system	1,315,712,234	1,823,962,385
Barnstable Great Marshes	280,851,338	455,834,605
Chase Garden Creek	113,078,364	183,994,859
Millway/Maraspin Creek	5,783,831	8,278,911

Residence times were averaged for the tidal cycles comprising a representative 14.5 tidal-day period (28 tide cycles), and are listed in Table V-12. The modeled time period used to compute the flushing rates correspond to the model calibration period, and included a full cycle of neap to spring tide conditions. The RMA-2 model calculated flow crossing specified grid continuity lines (similar to an ADCP transect) for each sub-embayment to compute the tidal prism volume. Since the half-lunar-month period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system sub-embayments.

Table V-12. Computed System and Local residence times for the Barnstable Great Marsh estuary system.

Embayment	System Residence Time (days)	Local Residence Time (days)
Barnstable Harbor/Chase Garden Cr. system	0.4	0.4
Barnstable Great Marshes	0.3	1.5
Chase Garden Creek	0.3	3.7
Millway/Maraspin Creek	0.4	82.2

The computed flushing rates for the entire system show that as a whole, the system flushes very well. A flushing time of 0.4 days for the entire estuary shows that on average, water is resident in the system for less than one half day. The low local residence times for the whole of the Barnstable Great Marsh estuary system show that water quality in the system is not impacted negatively by tidal flushing. This is a typical result for estuaries dominated by marsh resources or with extensive tidal flats, where the tide prism volume is of a magnitude comparable to the mean volume of the system.

For the smallest sub-embayments of the Harbor system, computed system residence times are typically two orders of magnitude longer than their corresponding local residence time. System residence times provide a qualitative measure that helps to identify the relative sensitivity of different sub-embayments to nutrient loading.

Based on our knowledge of estuarine processes, we estimate that the combined errors associated with the method applied to compute residence times are within 10% to 15% of “true” residence times, for the Barnstable Great Marsh estuary system. Possible errors in computed residence times can be linked to two sources: the bathymetry information and simplifications employed to calculate residence time. In this study, the most significant errors associated with the bathymetry data result from the process of interpolating the data to the finite element mesh, which was the basis for all the flushing volumes used in the analysis. In addition, limited topographic measurements were available in some of the smaller sub-embayments of the system.

Minor errors may be introduced in residence time calculations by simplifying assumptions. Flushing rate calculations assume that water exiting an estuary or sub-embayment does not return on the following tidal cycle. For regions where a strong littoral drift exists, this assumption is valid. However, water exiting a small sub-embayment on a relatively calm day may not completely mix with estuarine waters. In this case, the “strong littoral drift” assumption would lead to an under-prediction of residence time. Since littoral drift along the shoreline of Cape Cod Bay typically is strong because of the effects of the local winds and tidal induced mixing, the “strong littoral drift” assumption will cause only minor errors in residence time calculations.

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

Several different data types and calculations are required to support the water quality modeling effort for the Barnstable Great Marsh estuary system. These include the output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds, measurements of internal nitrogen loads from the sediment (benthic flux), and measurements of nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayments

Extensive field measurements and hydrodynamic modeling of the embayment was an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated hydrodynamic model representing the transport of water within the Barnstable Great Marsh estuary system. Files of node locations and node connectivity for the RMA-2V model grids were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model was also the computational grid for the water quality model. The period of hydrodynamic model output used for the water quality model calibration was the half-lunar-month (28 tide cycle) period beginning July 22, 2015 0430 EDT. This period overlaps with the time period used for the hydrodynamic model calibration and also the flushing analysis presented in Section V. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model to reach a dynamic “steady state”, and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayments

Three primary nitrogen loads to the Barnstable Harbor embayment were utilized in this modeling study: external loads from the watersheds, nitrogen load from direct rainfall on the embayment surface, and internal loads from the sediments. Additionally, there is a fourth load to the Barnstable Great Marsh estuary system, consisting of the background concentrations of total nitrogen in the water entering from Cape Cod Bay. This load is represented as a constant concentration along the seaward boundary of the model grid.

VI.1.3 Measured Nitrogen Concentrations in the Embayment

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in the area map presented in Figure VI-1. The multi-year averages present the “best” comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data are the minimum required to provide a baseline for MEP analysis. For stations in Barnstable Great Marsh estuary system, a minimum of seven years and up to 13 years of water quality data were available.

Table VI-1. Measured data and modeled bioactive nitrogen concentrations for the Barnstable Harbor estuarine system used in the model calibration plots of Figures VI-2 and VI-3. All concentrations are given in mg/L N. “Data mean” values are calculated as the average of all measurements. Data represented in this table were collected in the summers of 2002 through 2014. Not all stations have data from all 13 years.

Location	Monitoring station	Data Mean	s.d. all data	N	model min	model max	model average
Scorton Creek	BM-13	0.189	0.056	33	0.063	0.183	0.121
Spring Creek	BM-11	0.190	0.052	33	0.064	0.185	0.127
Barnstable Harbor – upper	BM-12	0.165	0.049	37	0.062	0.167	0.111
Barnstable Harbor – upper	BM-1	0.131	0.039	131	0.062	0.156	0.097
Barnstable Harbor – mid	BM-2	0.111	0.051	137	0.063	0.148	0.085
Barnstable Harbor – lower	BM-3	0.098	0.042	129	0.063	0.118	0.072
Broad Sound	BM-10	0.269	0.097	24	0.070	0.136	0.105
Bass Hole	BSH-1	0.107	0.023	41	0.062	0.187	0.093
Chase Garden Creek - lower	BSH-2	0.108	0.025	41	0.064	0.219	0.111
Chase Garden Creek	BSH-3	0.125	0.033	43	0.064	0.240	0.122
Chase Garden Creek	BSH-4	0.129	0.040	45	0.064	0.277	0.132
Chase Garden Creek	BSH-5	0.275	0.153	41	0.085	0.520	0.271
Chase Garden Creek – upper	BSH-6	0.767	0.206	40	0.383	0.834	0.571
Whites Brook	BSH-7	0.355	0.163	32	0.109	0.587	0.321

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in the Barnstable Harbor estuarine system. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of Barnstable Harbor. Like the RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. The MEP Technical Team has utilized this model in water quality studies of other embayment systems in southeastern Massachusetts, including Pleasant Bay (Howes *et al*., 2006); New Bedford Harbor (Howes *et al*., 2008); Edgartown Great Pond, MA (Howes *et al*., 2008) and Sandwich Harbor (Howes et al., 2014).

The overall approach involves modeling total nitrogen as a non-conservative constituent, where bottom sediments act as a source or sink of nitrogen, based on local biochemical characteristics. This modeling represents summertime conditions, when algal growth is at its maximum. Total nitrogen modeling is based upon various data collection efforts and analyses presented in previous sections of this report. Nitrogen loading information was derived from the MEP Technical Team watershed loading analysis, as well as the measured bottom sediment nitrogen fluxes. Water column nitrogen measurements were utilized as model boundaries and as calibration data. Hydrodynamic model output (discussed in Section V) provided the remaining information (tides, currents, and bathymetry) needed to parameterize the water quality model of the Barnstable Great Marsh estuary system.

Figure VI-1. Estuarine water quality monitoring station locations in the Barnstable Great Marsh estuary system. Station labels correspond to those provided in Table VI-1.

VI.2.1 Model Formulation

The formulation of the model is for two-dimensional depth-averaged systems in which concentration in the vertical direction is assumed uniform. The depth-averaged assumption is justified since vertical mixing by wind and tidal processes prevent significant stratification in the modeled embayment. The governing equation of the RMA-4 constituent model can be most simply expressed as a form of the transport equation, in two dimensions:

$$\left(\frac{\partial c}{\partial t} + u\frac{\partial c}{\partial x} + v\frac{\partial c}{\partial y}\right) = \left(\frac{\partial}{\partial x}D_x\frac{\partial c}{\partial x} + \frac{\partial}{\partial y}D_y\frac{\partial c}{\partial y} + \sigma\right)$$

where c is the water quality constituent concentration; t is time; u and v are the velocities in the x and y directions, respectively; D_x and D_y are the model dispersion coefficients in the x and y directions; and σ is the constituent source/sink term. Since the model utilizes input from the RMA-2 model, a similar implicit solution technique is employed for the RMA-4 model.

The model is therefore used to compute spatially and temporally varying concentrations c of the modeled constituent (i.e., total nitrogen), based on model inputs of: 1) water depth and velocity computed using the RMA-2 hydrodynamic model; 2) mass loading input of the modeled constituent; and 3) user selected values of the model dispersion coefficients. Dispersion coefficients used for each system sub-embayment were developed during the calibration process. During the calibration procedure, the dispersion coefficients were incrementally changed until model concentration outputs matched measured data.

The RMA-4 model can be utilized to predict both spatial and temporal variations in total for a given embayment system. At each time step, the model computes constituent concentrations over the entire finite element grid and utilizes a continuity of mass equation to check these results. Similar to the hydrodynamic model, the water quality model evaluates model parameters at every element at 10-minute time intervals throughout the grid system. For this application, the RMA-4 model was used to predict tidally averaged total nitrogen concentrations throughout the Barnstable Great Marsh estuary system.

VI.2.2 Water Quality Model Setup

Required inputs to the RMA-4 model include a computational mesh, computed water elevations and velocities at all nodes of the mesh, constituent mass loading, and spatially varying values of the dispersion coefficient. Because the RMA-4 model is part of a suite of integrated computer models, the finite-element meshes and the resulting hydrodynamic simulations previously developed for Barnstable Harbor also were used for the water quality constituent modeling portion of this study.

For each model, an initial total N concentration equal to the concentration at the open boundary was applied to the entire model domain. The model was then run for a simulated month-long (28 day) spin-up period. At the end of the spin-up period, the model was run for an additional 14 day (336 hour) period. Model results were recorded only after the initial spin-up period. The time step used for the water quality computations was 10 minutes, which corresponds to the time step of the hydrodynamics input for the Barnstable Harbor hydrodynamic model.

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included: 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, and 3) summer benthic regeneration. Nitrogen loads from each separate sub-watershed to the embayment were distributed by watershed. For example, the watershed load for the Millway was input within the model area that represents this small sub-embayment of the Harbor. Benthic loads were distributed between the grid elements within the separate tidal creeks in the Harbor system.

The loadings used to model present conditions in the Barnstable Great Marsh estuary system are given in Table VI-2. Watershed and depositional loads were taken from the results of the analysis of Section IV. Summertime benthic flux loads were computed based on the analysis of sediment cores in Section IV. The area rate (g/sec/m^2) of nitrogen flux from that analysis was applied to the surface area coverage computed for each sub-embayment (excluding marsh coverages, when present), resulting in a total flux for each portion of the overall embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. Fluxes range between negative (uptake of nitrogen) and positive (source of nitrogen) values in different areas of the system. In the main Harbor basin, the net benthic flux is negative which indicates a net uptake of nitrogen in the bottom sediments. The greatest measured positive fluxes exist in the Millway basin, the dredged boat basin in Barnstable Village.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. The bioactive nitrogen concentration of the incoming water was set at the value designated for the open boundary. The boundary concentration in Cape Cod Bay, offshore the harbor inlet, was set at 0.063 mg/L, based on SMAST data collected offshore in the Bay.

Table VI-2. Sub-embayment and surface water loads used for total nitrogen modeling of the Barnstable Great Marsh estuary system, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent **present loading conditions** for the listed sub-embayments.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Barnstable Harbor - west	38.586	2.422	5.304
Barnstable Harbor - mid	27.562	13.800	-11.497
Barnstable Harbor - east	42.600	32.978	-23.366
Millway	10.575	0.211	2.853
Chase Garden Creek - west	29.666	0.140	0.373
Chase Garden Creek - east	25.030	0.003	-
Bass Hole	7.408	0.488	-1.079
System Total	181.427	50.041	-27.412

VI.2.4 Model Calibration

The development of the Barnstable Harbor water quality model began with the parameterization and calibration of the salinity model. Salinity is a conservative water quality constituent and therefore ideally suited for model calibration. Model dispersion coefficients were adjusted so that model output salinity matched measured data from the harbor. Generally, several model runs were required to bring the model into agreement with the water column measurements. Dispersion coefficient (E) values were varied through the modeled system by setting different values of E for each grid material type, as designated in Section V. Observed values of E in coastal estuary areas typically range between order 10 and order 0.001 m^2/sec (USACE, 2001). The final values of E used in each sub-embayment of the modeled system are presented in Table VI-3. These values were used to develop the "best-fit" salinity model calibration. For the case of salinity modeling, "best fit" can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within the model domain.

The only required inputs into the RMA-4 salinity model of the system, in addition to the RMA-2 hydrodynamic model output, were salinities at the model open boundary, and freshwater inputs (including inputs from rain, surface streams and groundwater). The open boundary salinity in Cape Cod Bay was set at 31.1 ppt. Surface water and groundwater input salinities were set at 0 ppt. Fresh water inputs into the model are listed in Table VI-4, which includes the sum of groundwater, surface streams, and direct rainfall onto the open water estuary surface of the Harbor, for each model sub-division.

Comparisons between calibrated model output and measured salinity are shown in plots presented in Figures VI-2 and VI-3. In these plots, means of the water column data and a range of two standard deviations of the annual means at each individual station are plotted against the modeled maximum, mean, and minimum concentrations output from the model at locations which corresponds to the MEP monitoring stations.

For model calibration, the target modeled salinities were compared to mean measured salinity data values at all water-quality monitoring stations. The calibration target was set between the modeled maximum and tidal averaged concentration at each station, in order to represent samples collected at or after the time of mid-ebb tide offshore in Cape Cod Bay.

Table VI-3. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for the Barnstable Great Marsh estuary system.

Embayment Division	E m^2/sec
Chase Garden Creek marsh	10.0
Bass Hole	10.0
Lone Tree Creek marsh plain	1.0
Barnstable Harbor main basin	1.0
Barnstable Harbor inlet	5.0
Barnstable Harbor marsh plain	0.5
Barnstable Harbor marsh creeks	0.5
Millway	1.0
Maraspin Creek	1.0

Table VI-4. Freshwater inputs, including groundwater, surface water (streams), and direct rainfall on estuary surface used as inputs to the model of the Barnstable Great Marsh estuary system.

estuary subdivision/stream input	flow ft^3/sec
Great Marsh West Barnstable	15.68
Alder Creek	0.39
Boat Cove Creek	5.14
Bridge Creek	2.32
Great Marsh Mid	18.04
Great Marsh BarnYarm	17.99
Huckins Neck Total	1.60
Millway Total	2.06
Maraspin Creek	0.57
Chase Garden Creek Salt W	2.97
Whites Brook Salt	2.52
Chase Garden Creek Salt E	3.61
Bass Hole	3.60

Also presented in Figure VI-3 are unity plot comparisons of measured data verses modeled target values for each system. The computed R^2 correlation is 0.92 and the root mean squared (rms) error is 3.2 ppt, both of which demonstrate good agreement between modeled and measured data for this system.

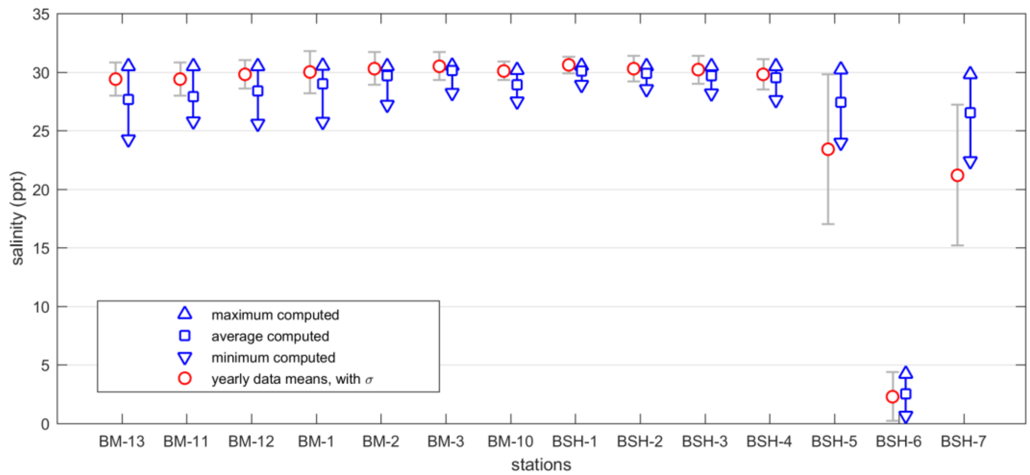


Figure VI-2. Comparison of measured total salinity and calibrated model output at stations in the Barnstable Great Marsh estuary system. Station labels correspond with the MEP IDs provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset

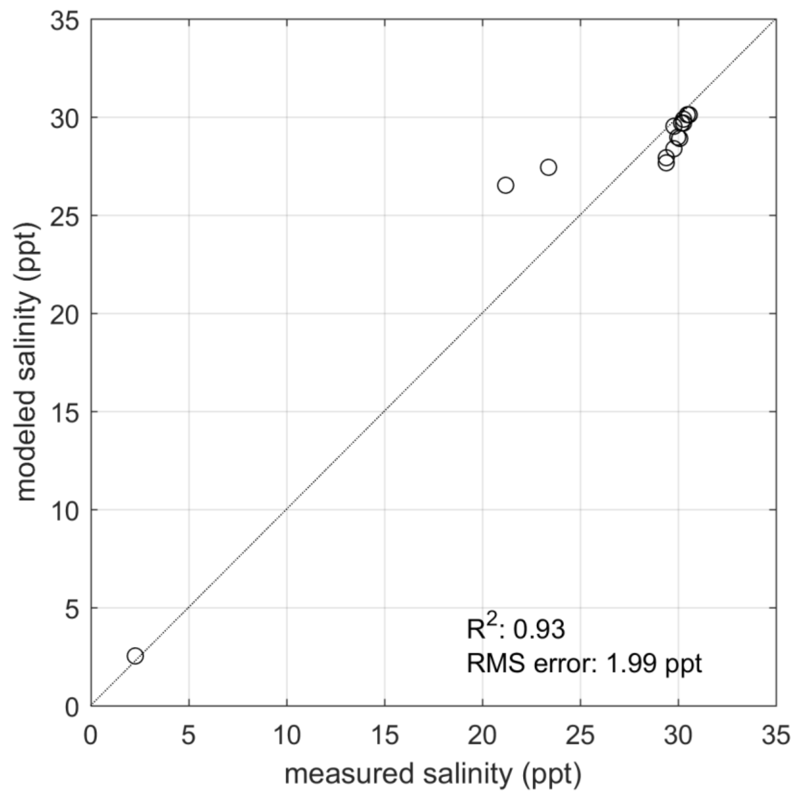


Figure VI-3. Model salinity calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for the model are 0.93 and 1.99 ppt respectively.

A contour plot of calibrated model output is shown in Figure VI-4. In this figure, color contours indicate salinity throughout the model domain. The output in the figure shows average salinity concentrations, computed using the full 14-tidal-day model simulation output period.

VI.2.5 Model Verification

In addition to the model calibration based on salinity, the numerical water quality model performance was verified by modeling bioactive nitrogen (bioactive N). This step was performed for the Barnstable Great Marsh estuary system using N measurements collected at the same stations as the salinity data and N loads from Table VI-2. For the bioactive N verification, none of the model dispersion coefficients were changed from the values used in the salinity calibration. Comparisons of modeled and measured N concentrations are presented in Figures VI-5 and VI-6, with contour plots of model output shown in Figure VI-7. The R^2 correlation of the model and measurements is 0.95 and the rms error of the model is 0.037 mg/L.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

To assess the influence of nitrogen loading on total nitrogen concentrations within the Barnstable Harbor, the standard "build-out" and "no-load" water quality modeling scenarios were run. These runs included a "build-out" scenario, based on potential development (described in more detail in Section IV), and a "no anthropogenic load" or "no load" scenario assuming only atmospheric deposition on the watershed and sub-embayment, as well as a natural forest within each watershed. Comparisons of the alternate watershed loading analyses are shown in Table VI-5. Loads are presented in kilograms per day (kg/day) in this Section, since it is inappropriate to show benthic flux loads in kilograms per year due to seasonal variability.

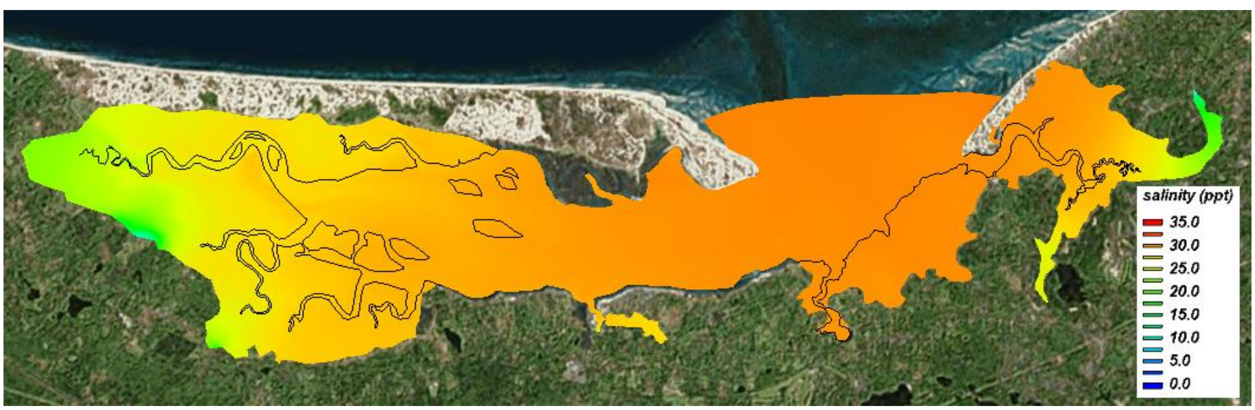


Figure VI-4. Contour Plot of average modeled salinity (ppt) in the Barnstable Great Marsh estuary system.

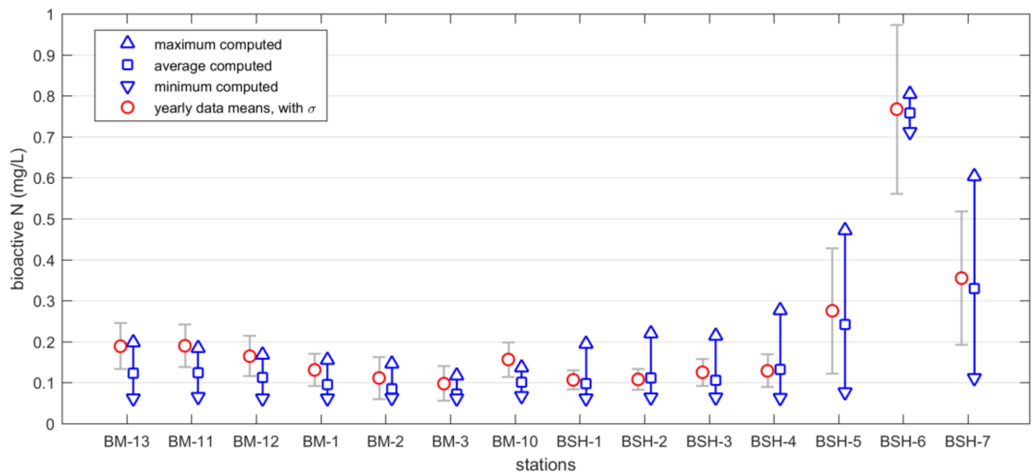


Figure VI-5. Comparison of measured and calibrated TN model output at stations in Barnstable Harbor. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed TN concentrations for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset.

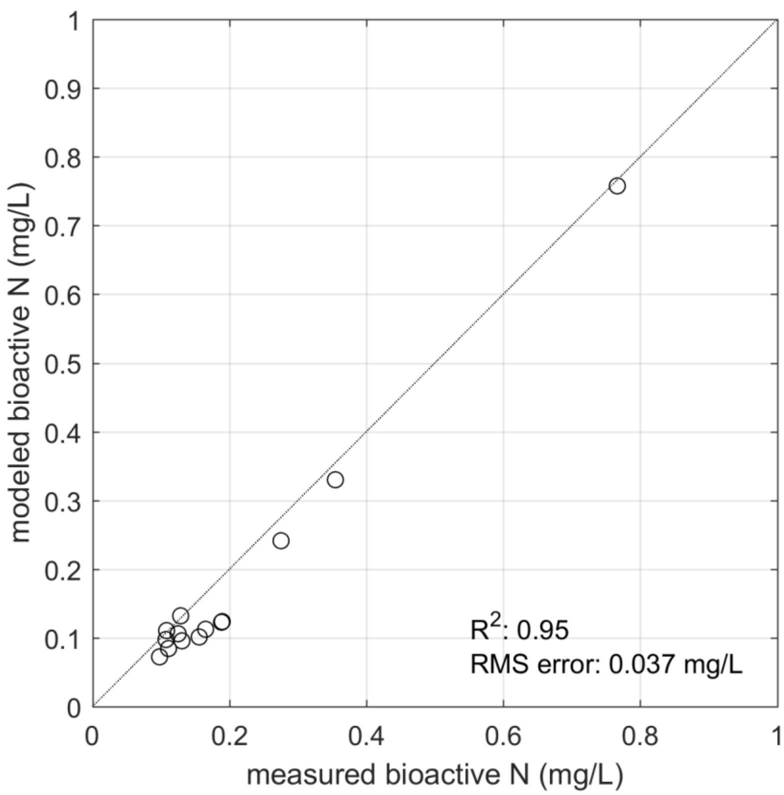


Figure VI-6. Model TN target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) is 0.95 and RMS error for this model verification run is 0.037 mg/L.

Figure VI-7. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for the Barnstable Great Marsh estuary system

Table VI-5. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of the Barnstable Great Marsh estuary system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	Build-out (kg/day)	build-out % change	no load (kg/day)	no load % change
Barnstable Harbor - west	38.586	48.452	+25.6%	5.518	-85.7%
Barnstable Harbor - mid	27.562	33.964	+23.2%	3.159	-88.5%
Barnstable Harbor - east	42.600	60.115	+41.1%	3.458	-91.9%
Millway	10.575	12.679	+19.9%	1.800	-83.0%
Chase Garden Creek - west	29.666	31.896	+7.5%	1.301	-95.6%
Chase Garden Creek - east	25.030	27.581	+10.2%	0.567	-97.7%
Bass Hole	7.408	7.770	+4.9%	0.564	-92.4%
System Total	181.427	222.458	+22.6%	13.934	-92.3%

VI.2.6.1 Build-Out

A breakdown of the total nitrogen load entering each sub-embayment is shown in Table VI-6 for the modeled build-out scenario. The benthic flux for the build-out scenarios is assumed to vary proportional to the watershed load, where an increase in watershed load will result in an increase in benthic flux (i.e., a positive change in the absolute value of the flux), and *vice versa*.

Projected benthic fluxes (for both the build-out and no load scenarios) are based upon projected PON concentrations and watershed loads, determined as:

$$(Projected\ N\ flux) = (Present\ N\ flux) * [PON_{projected}]/[PON_{present}]$$

where the projected PON concentration is calculated by,

$$[PON_{projected}] = R_{load} * \Delta PON + [PON_{(present\ offshore)}],$$

using the watershed load ratio,

$$R_{load} = (Projected\ N\ load)\ /\ (Present\ N\ load),$$

and the present PON concentration above background,

$$\Delta PON = [PON_{(present\ flux\ core)}] - [PON_{(present\ offshore)}].$$

Following development of the nitrogen loading estimates for the build-out scenario, the water quality models of the system was run to determine nitrogen concentrations within each sub-embayment (Table VI-7). In this table, the percent change *P* over background presented in this table is calculated as:

$$P = (N_{scenario}-N_{present})/(\ N_{present}-N_{background})$$

where N is the nitrogen concentration at the indicated monitoring station for present conditions and the loading scenario (i.e., build-out in this case), and also in Cape Cod Bay (background). Bioactive nitrogen concentrations in the receiving waters (i.e., Cape Cod Bay) remained identical to the existing conditions modeling scenarios. For build-out, the percent increase in modeled TN concentrations is greatest at the station near the Harbor mouth (BM-3) and at Broad Sound in the mid-harbor region (BM-10). Concentrations increased 25% above background at these two monitoring stations. The largest bioactive N magnitude change occurs at station BSH-6 in upper Chase Garden Creek, where average bioactive N increases 0.054 mg/L. A contour plot showing average TN concentrations throughout the harbor system is presented in Figure VI-8 for the model of build-out loading.

Table VI-6. **Build-out** scenario sub-embayment and surface water loads used for total nitrogen modeling of the Barnstable Great Marsh estuary system, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Barnstable Harbor - west	48.452	2.422	5.862
Barnstable Harbor - mid	33.964	13.800	-11.903
Barnstable Harbor - east	60.115	32.978	-22.511
Millway	12.679	0.211	3.092
Chase Garden Creek - west	31.896	0.140	0.383
Chase Garden Creek - east	27.581	0.003	-
Bass Hole	7.770	0.488	-1.136
System Total	222.458	50.041	-26.213

Table VI-7. Comparison of model average bioactive N concentrations from present loading and the **build-out scenario**, with percent change over background in Cape Cod Bay (0.063 mg/L), for the Barnstable Great Marsh estuary system.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	build-out (mg/L)	% change
Scorton Creek	BM-13	0.121	0.136	+24.7%
Spring Creek	BM-11	0.127	0.141	+23.0%
Barnstable Harbor – upper	BM-12	0.111	0.123	+24.0%
Barnstable Harbor – upper	BM-1	0.097	0.105	+24.2%
Barnstable Harbor – mid	BM-2	0.085	0.090	+23.5%
Barnstable Harbor – lower	BM-3	0.072	0.075	+25.5%
Broad Sound	BM-10	0.105	0.116	+25.0%
Bass Hole	BSH-1	0.093	0.095	+8.8%
Chase Garden Creek - lower	BSH-2	0.111	0.115	+8.6%
Chase Garden Creek	BSH-3	0.122	0.127	+9.0%
Chase Garden Creek	BSH-4	0.132	0.139	+8.9%
Chase Garden Creek	BSH-5	0.271	0.292	+9.9%
Chase Garden Creek – upper	BSH-6	0.571	0.626	+10.6%
Whites Brook	BSH-7	0.321	0.344	+8.6%

Figure VI-8. Contour plot of modeled total nitrogen concentrations (mg/L) in the Barnstable Great Marsh estuary system, for projected build-out scenario loading conditions.

VI.2.6.2 No Anthropogenic Load

A breakdown of the total nitrogen load entering each sub-embayment for the no anthropogenic load ("no load") scenarios is shown in Table VI-8. The benthic flux input to each embayment was reduced (toward zero) based on the reduction in the watershed load (as discussed in Section VI.2.6.1). Compared to the modeled present conditions and build-out scenario, atmospheric deposition directly to each sub-embayment becomes a greater percentage of the total nitrogen load as the watershed load and related benthic flux decrease.

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations at each monitoring station. Again, total nitrogen concentrations in the receiving waters (i.e., Cape Cod Bay) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from “no load” was large, with all areas of the system experiencing reductions greater than 100%, compared to the background concentration of 0.063 mg/L in Cape Cod Bay (Table VI-9). Generally, average bioactive N concentrations drop from present conditions. The greatest drop occurs in Chase Garden Creek. One station near the Harbor entrance (BM-3) has an increased concentration, which results from a decrease in the negative benthic flux between the present and “no load” scenarios. This decrease in negative flux causes a reduction of uptake of nitrogen by the Harbor bottom sediments which leads to an increase in N concentrations in this area of the harbor. A contour plot showing TN concentrations throughout the system is presented in Figure VI-9.

Table VI-8. “**No anthropogenic loading**” (“no load”) sub-embayment and surface water loads used for total nitrogen modeling of the Barnstable Great Marsh estuary system, with total watershed N loads, atmospheric N loads, and benthic flux

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Barnstable Harbor - west	5.518	2.422	3.210
Barnstable Harbor - mid	3.159	13.800	-10.084
Barnstable Harbor - east	3.458	32.978	-23.655
Millway	1.800	0.211	1.902
Chase Garden Creek - west	1.301	0.140	0.306
Chase Garden Creek - east	0.567	0.003	-
Bass Hole	0.564	0.488	-0.937
System Total	16.367	50.041	-29.258

Table VI-9. Comparison of model average bioactive N concentrations from present loading and the “**No anthropogenic loading**” (“no load”), with percent change over background in Cape Cod Bay (0.063 mg/L), for the Barnstable Great Marsh estuary system.

Station Location	monitoring station (MEP ID)	present (mg/L)	“no load” (mg/L)	% change
Scorton Creek	BM-13	0.121	0.086	-60.0%
Spring Creek	BM-11	0.127	0.091	-56.4%
Barnstable Harbor – upper	BM-12	0.111	0.087	-51.4%
Barnstable Harbor – upper	BM-1	0.097	0.084	-39.2%
Barnstable Harbor – mid	BM-2	0.085	0.083	-10.9%
Barnstable Harbor – lower	BM-3	0.072	0.080	+78.7%
Broad Sound	BM-10	0.105	0.087	-44.3%
Bass Hole	BSH-1	0.093	0.074	-64.2%
Chase Garden Creek - lower	BSH-2	0.111	0.074	-76.8%
Chase Garden Creek	BSH-3	0.122	0.074	-81.6%
Chase Garden Creek	BSH-4	0.132	0.074	-84.3%
Chase Garden Creek	BSH-5	0.271	0.072	-95.6%
Chase Garden Creek – upper	BSH-6	0.571	0.052	-102.2%
Whites Brook	BSH-7	0.321	0.077	-94.7%

Figure VI-9. Contour plot of modeled total nitrogen concentrations (mg/L) in Barnstable Harbor, for no anthropogenic loading conditions.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Barnstable Great Marsh estuary system in the Towns of Barnstable and Dennis, MA, the assessment is based upon data from water quality monitoring conducted by the Towns of Barnstable and Dennis with technical support from the Coastal Systems Program (UMASS-SMAST), MassDEP surveys of eelgrass distribution as available (typically 1951, 1995, 2001 and checks by SMAST in 2007), benthic animal communities (fall 2007), sediment characteristics (summer 2007), and dissolved oxygen records (summer 2007). Water quality monitoring in Barnstable Harbor was initiated in 2002 at stations BM1,2,3 and has persisted till present. In 2008 four additional stations were added to the program, BM10,11,12,13 and sampling of these stations continues to present as well. Water quality sampling in the Great Marshes portion of the estuary has also been periodically undertaken at 5 additional stations (GM1,2,3,4,5) in 2005, 2008 and 2012-2014. In Bass Hole, water quality sampling has been continuously conducted by the Town of Dennis at stations BH1-7 from 2005 to 2014. These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support nitrogen threshold development for this system (Section VIII). It should be noted that nitrogen enrichment occurs through 2 primary mechanisms, high rates of nitrogen entering from the surrounding watershed and/or low rates of flushing due to restriction of tidal exchange with the low nitrogen waters of Cape Cod Bay. The Barnstable Great Marsh estuary system has increasing nitrogen loading from the associated watersheds from shifting land-uses and may have periodic alterations in circulation and possibly tidal exchange in specific portions of the system due to the dynamics of sand in the main basin and in and around the mouth of Bass Hole. Fundamentally, restrictions of tidal exchange increase the sensitivity of an estuary to nitrogen inputs, however, this is more so the case in a classic embayment setting as opposed to systems like Barnstable Great Marsh estuary system that are dominated by salt marshes with higher assimilative capacities for nitrogen than open water embayments.

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

There are a variety of indicators that can be used in concert with water quality monitoring data for evaluating the ecological health of embayment systems. The best biological indicators are those species which are non-mobile and which persist over relatively long periods, if environmental conditions remain constant. The concept is to use species which integrate environmental conditions over seasonal to annual intervals. The approach is particularly useful in environments where high-frequency variations in structuring parameters (e.g. light, nutrients, dissolved oxygen, etc.) are common, making adequate field sampling difficult.

As a basis for a nitrogen thresholds determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll-*a* (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed dissolved oxygen sensors at strategic locations throughout the Barnstable Great Marsh estuary system to record the frequency and duration of low oxygen conditions during the critical summer period. The MEP habitat analysis uses eelgrass as a sentinel species for indicating nitrogen over-loading to coastal embayments. Eelgrass loss in Cape Cod estuaries associated with nitrogen enrichment is generally through decreased light penetration resulting from increased phytoplankton biomass and resulting suspended organic particles, as well as shading by epiphytes (small plants that colonize eelgrass shoots) and sometimes by drift macroalgae. Each of these factors is a result of nitrogen enrichment and all result in stress to eelgrass beds.

MassDEP mapping of the eelgrass beds was not conducted within the component basins of the Barnstable Great Marsh estuary system, although some surveys were conducted in the near shore waters of Cape Cod Bay just outside the tidal inlet (MassDEP Eelgrass Mapping Program, C. Costello 2010-2013). However, a historical analysis of possible eelgrass distribution was conducted within the estuary which showed only limited area. This is likely because many of the central basin areas have very dynamic sediments, with unstable sands that do not support eelgrass. Surveying completed by the SMAST-MEP Technical Team in the summer of 2007 confirmed the absence of any eelgrass, as would be expected in a tidal salt marsh dominated system that is also composed of a large open water area with a large tidal range, strong tidal currents and large areas of shifting sand flats and sand waves. As a result, temporal changes in eelgrass distribution could not provide a basis for evaluating recent increases (nitrogen loading) in nutrient enrichment of the Barnstable Great Marsh estuary system. The low phytoplankton and macroalgal biomass, oxidized sediments and high light penetration is consistent with a non-nitrogen factor causing the absence of eelgrass, most likely associated with high tidal velocities, unstable sediments and possibly winter storm exposure. As a result, nutrient threshold determination was based strongly on results from the dissolved oxygen and chlorophyll mooring data as well as the benthic infaunal community characterization. Prior to evaluating nitrogen thresholds, analysis of inorganic N/P molar ratios was conducted within the water column throughout the estuary comprised of Barnstable Great Marsh estuary system and their tributary creeks. The results support the contention that nitrogen is the nutrient to be managed in this estuary, as the N/P molar ratio throughout this large estuarine complex averaged 4.6 with a range of 3.1-7.4, clearly below the Redfield Ratio value (16) indicating that nitrogen additions will increase phytoplankton production in this estuary.

In areas that do not support eelgrass beds, benthic animal indicators were used to assess the level of habitat health from “healthy” (low organic matter loading, high D.O.) to “highly stressed” (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stressed indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, benthic population studies in Buzzards Bay (Woods Hole Oceanographic Institution) and New Bedford (SMAST), and more recently the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes *et al.* 1997). These data are coupled with the level of diversity (H’) and evenness (E) of the benthic community and the total number of individuals to determine the infaunal habitat quality.

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, USEPA[1] suggests that the chronic protective oxygen level to support growth of estuarine animals is 4.8 mg L^{-1}, with a limit for survival of juvenile and adult organisms of 2.3 mg L^{-1}. However, studies have demonstrated that slightly higher oxygen levels, 3.0 mg/L, can be lethal to larval fish and crustaceans (Poucher and Coiro 1997). Massachusetts State Water Quality Classification indicates that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the Barnstable Great Marsh estuary system are currently listed under this classification as SA. It should be noted that the classification system represents the water quality that the embayment should support, not the existing level of water quality. It is through the MEP and TMDL processes that management actions are developed and implemented to keep or bring the existing conditions in line with the classification.

Dissolved oxygen levels in temperate estuaries vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (Figure VII-1). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute oxygen levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the bottom of the main tidal creeks within key regions of the Barnstable Great Marsh estuary system (Figure VII-2). The dissolved oxygen sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployments. In addition periodic calibration samples were collected at the depth of each sensor and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 30 days within the interval from July through mid-September. All of the mooring data from the Barnstable Great Marsh estuary system were collected during the summer of 2007.

Similar to other estuaries in southeastern Massachusetts, the Barnstable Great Marsh estuary system evaluated in this assessment showed high frequency variation related primarily to diurnal influences and to tidal influences. Nitrogen enrichment of estuarine waters generally manifests itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. The high degree of temporal variation in bottom water dissolved oxygen concentration at each mooring site, underscores the need for continuous monitoring within these systems.

[1] USEPA 2000. Ambient Aquatic Life Water Quality Criteria for Dissolved Oxygen (Saltwater): Cape Cod to Cape Hatteras (133 p.).

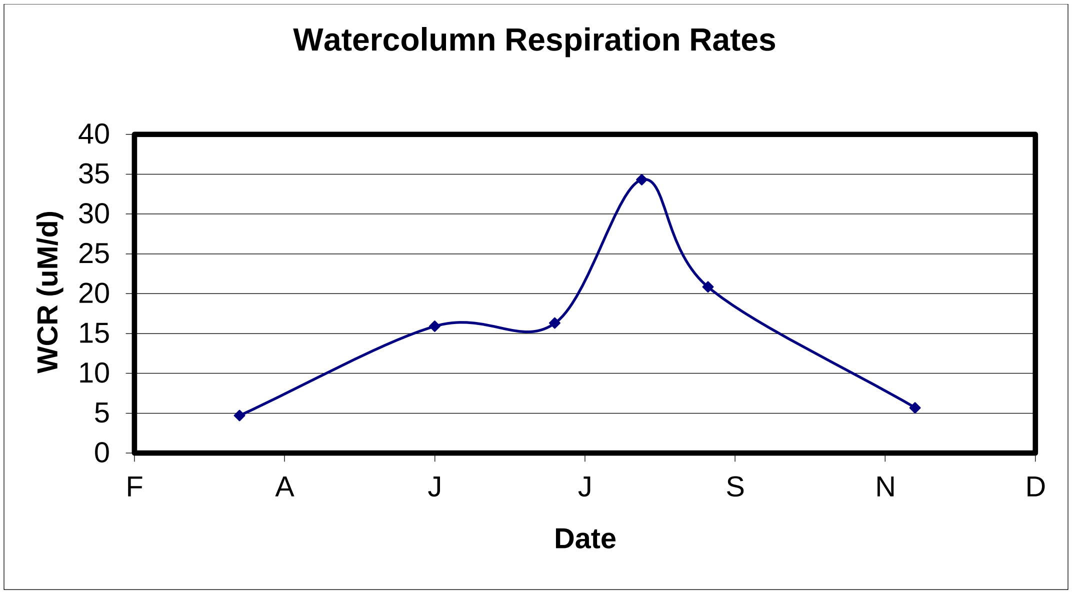


Figure VII-1. Example of typical average water column respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System, Cape Cod (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

Dissolved oxygen and chlorophyll-*a* records were examined both for temporal trends and to determine the percent of the 28-36 day deployment period that these parameters were below/above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms.

The oxygen data in the tributary creeks is consistent with high organic matter inputs from the surrounding vegetated marsh and creek banks rather than from phytoplankton which show relatively low to moderate enrichment (chlorophyll-a levels generally <10 ug L-1). The high velocity of tidal currents in the main tidal channels (e.g. Great Marshes Main Basin, Scorton Creek, Spring Creek, Bass Hole Upper and Lower) and near complete draining of the creeks at low tide appears to reduce the settling of phytoplankton, hence sediment oxygen uptake. An indication of the areas with low organic matter deposition is the large swept sand areas and sand waves with low organic content and low rates of oxygen uptake. That large portions of the Barnstable Great Marsh estuary system are dominated by salt marsh habitat as well as the large tidal range and vigorous flushing are factors that must be taken into consideration when interpreting the dissolved oxygen data and its relation to nutrient enrichment of the system and any determination of habitat impairment. The observed levels of oxygen depletion in this large estuary and the magnitude of daily oxygen excursion and chlorophyll-*a* levels are consistent with the natural organic rich salt marsh creeks with high tidal flushing that are not being overly enriched by watershed inputs (Figures VII-3 through VII-14).

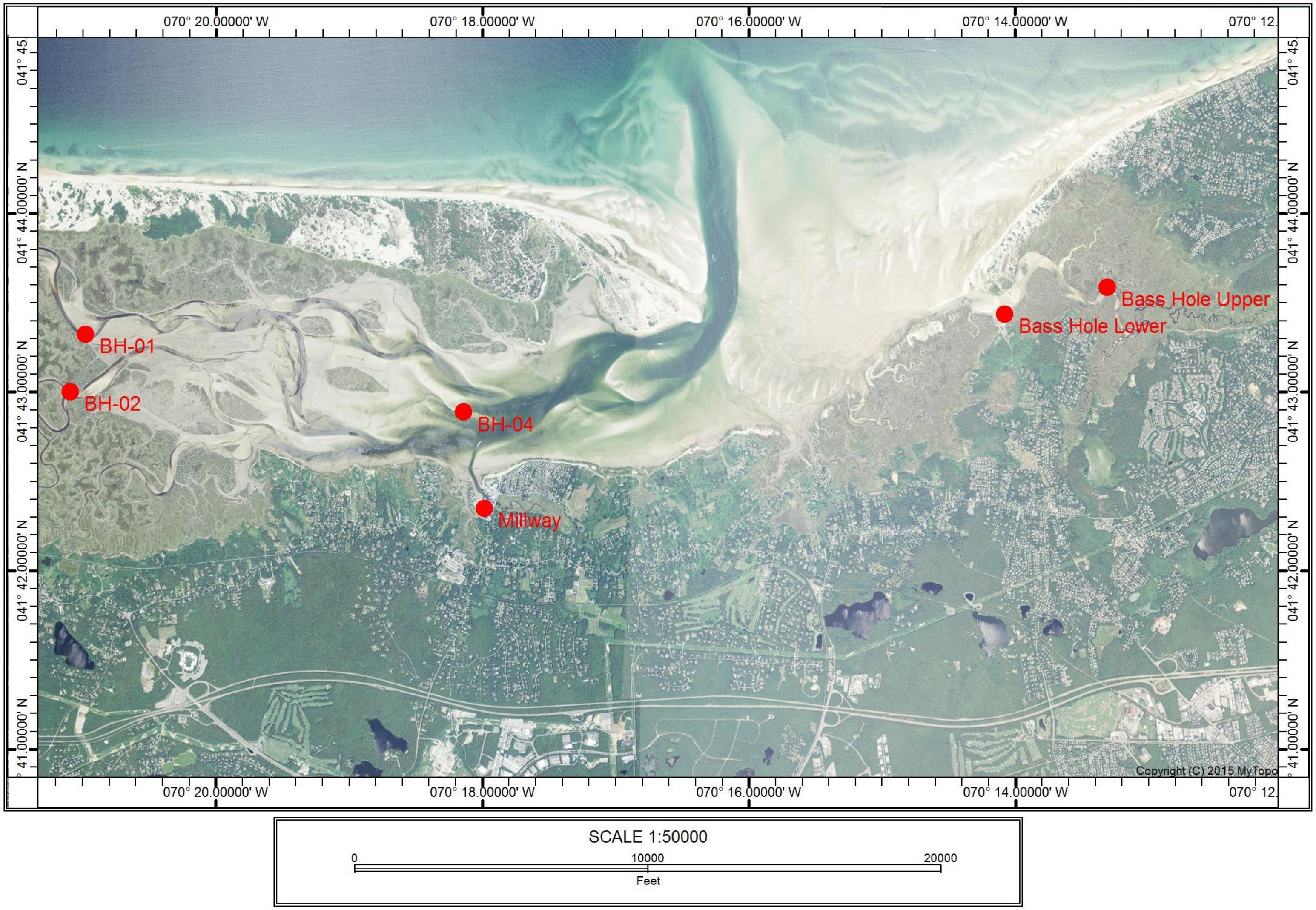


Figure VII-2. Aerial Photograph of the Barnstable Great Marsh estuary system in the Towns of Barnstable and Dennis showing the location of the continuously recording Dissolved Oxygen / Chlorophyll-*a* sensors deployed during the Summer of 2007. BH-01 = Scorton Creek, BH-02 = Springhill Creek, BH-04 = Blish Point (middle of central main basin).

Interpretation of estuarine oxygen records need to consider both the frequency of oxygen depletion and the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions. The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in a particular location. Nitrogen enrichment also results in increased phytoplankton (or epibenthic algae) production, as evidenced by oxygen levels that rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). In the Barnstable Great Marsh estuary system, oxygen levels only significantly exceeded atmospheric equilibrium occasionally, although oxygen depletion was commonly observed. The absence of elevated oxygen levels is consistent with the oxygen variations being the response to the naturally organic rich nature of tidal salt marsh creeks rather than eutrophying levels of nitrogen. In these systems the oxygen dynamic is driven by consumption within the tidal creeks, with re-oxygenation through phytoplankton production being limited. As the creeks drain nearly completely at low tide, the rise in oxygen levels is primarily through the entry of oxygen rich coastal waters on the flooding tide.

The dissolved oxygen records indicate that in the nearby Scorton Creek Salt Marsh the upper tidal creeks (away from the tidal inlet) and lower regions of the central main tidal salt marsh creek frequently have oxygen declines to ~4 mg L^{-1}, while the tributary tidal creek along the backside of the barrier beach (Scorton Creek 2 mooring) shows periodic (but not prolonged) oxygen depletion to 2 mg L^{-1} (Table VII-1). Such oxygen depletion is typical of organic and nutrient rich temperate salt marsh creeks. Salt marshes are nutrient and organic matter enriched as part of their ecological design, which makes them such important nursery areas for adjacent offshore waters. However, a natural consequence of their organic rich sediments is periodic oxygen depletion within the tidal creeks, particularly during the summer. The observed level of oxygen depletion in the Scorton Creek salt marsh system is expected, as was the nearly identical pattern recorded by the MEP Technical Team in nearby Namskaket and Little Namskaket Creeks, both located on Cape Cod Bay in the Town of Orleans. Assessment of habitat quality must necessarily consider the natural function and tolerances of the specific estuarine ecosystems being evaluated. The specific results are as follows:

Barnstable Harbor-Bass Hole DO/CHLA Moorings (Figures VII-3 through VII-8):

The Barnstable Great Marsh estuary system is functionally a combination of a large open water main basin with a large tidal range and extensive sand flats and swept sands combined with large tidal salt marsh (the Great Marshes) that has several central tidal creeks and a large tributary salt marsh (Bass Hole, aka Chase Garden Creek) with a central tidal creek.. There are also a number of smaller salt marsh creeks tributary to the main open water basin. The upper reaches of the tidal creeks that penetrate into the salt marshes abutting the main open water basin have nearly vertical creek banks surrounded by extensive emergent marsh vegetated with typical New England high and low salt marsh plants. The lower reaches of the tidal creeks form broader creeks with sediments comprised of marsh deposits and sand transported in by coastal processes. The mid and lower portions of the central creeks have high tidal velocities as they drain water from the extensive upper marsh areas. As a result, the creek bottom sediments of the mid and lower reaches primarily consist of migrating fine and medium sands with some areas of coarse sand and gravel. The sides of the creeks consist of eroding salt marsh peat. In contrast, the upper creek areas support organic rich sediments with some fine sand mixed in. The tide range in adjacent Cape Cod Bay is large, ~10 ft (Chapter V), and the salt marsh areas are regularly flooded at high tide and the salt marsh creeks drain nearly completely with each ebb tide. This is significant when interpreting periodic low DO measurements at some of the mooring sites located in salt marsh dominated areas of this estuarine system.

Moderate to large diurnal shifts in dissolved oxygen were measured at each of the 6 mooring sites, but this is particularly evident in the four (4) moorings deployed in the main tidal creeks discharging to Barnstable Harbor (e.g. Scorton Creek, Spring Creek and Chase Garden Creek which is the main tidal creek that makes up Bass Hole. Interestingly, both the Scorton Creek and the Spring Creek moorings generally maintained oxygen levels >4 mg L^{-1}, which is relatively high for salt marsh creeks. It is likely that the near draining of the creek water at low tide and the large tidal flows with low nutrient and high oxygen Cape Cod Bay water plays a significant role in the oxygen balance, as does the low organic sandy sediments with only moderate levels of oxygen uptake (D. Schlezinger, personal communication). Overall, the oxygen conditions are generally higher than in smaller enclosed temperate salt marshes which are naturally organic and nutrient rich ecosystems. The chlorophyll-*a* levels are generally consistent with this conclusion, however, the average levels over 4 of the deployments were moderate, ranging from 10.2 - 15.2 ug L^{-1} for the two moorings located in the Great Marsh (Scorton Creek and Spring Creek) and 8.5 - 9.0 ug L^{-1} for the two moorings located in Bass Hole (lower and upper respectively. Equally important the Town Water Quality Monitoring Programs have long-term chlorophyll-*a* records from traditional sampling approaches with comparable averages of 3.9 – 6.5 for the main basin, 8.9 – 11.6 in the small tidal creeks of the Great Marshes along the southern shore of the main basin and 8.5 and 9.0 for lower and upper Bass Hole. The high tidal flushing does not generally allow a buildup of phytoplankton during the short residence time (generally < 1 tidal cycle) of phytoplankton within the tidal creeks, however, it is evident in that periodic blooms do occur mainly in the tributary tidal creeks as seen in the continuous records from Scorton Creek and the Spring Creek during the last third of the deployment period. A similar bloom event appears to be developing around the same point in time at the Blish Point mooring located in the mid central open water basin near the inlet to the Millway. Similarly, a bloom event was observed in the tributary marsh of Bass Hole though the upper and lower moorings in that salt marsh indicate that the bloom began earlier than in the Great Marshes, in the first two weeks of the deployment.

In regard to the dissolved oxygen records from the moorings deployed in the salt marsh creeks, the high velocities appear to reduce the sediment oxygen demand and promote sandy oxidized surface sediments over much of the mid and lower tidal reaches. However, it appears that transfers from the emergent marsh and from the upper to lower marsh are sufficient to create the observed periodic low oxygen levels such as those measured in Bass Hole and at times in Spring Creek. It should be noted that there were no prolonged (e.g. several day) hypoxic events in this system, as found in impaired open water basins and the degree of oxygen depletion was significantly less than observed in many healthy New England and Cape Cod salt marshes. Instead, oxygen levels generally cycled from atmospheric equilibration (7 - 8 mg L^{-1}) to ~ 4 to ~2 mg L^{-1} for all sites.

The low to moderate chlorophyll-*a* concentrations at all the mooring locations show modest enhancement over the offshore waters, as the near complete exchange of tidal waters on each tide in the creeks does not allow for chlorophyll levels to build. The absence of prolonged (i.e. multi tidal cycle) oxygen depletion, typically found in nitrogen enriched embayments (due to stimulation of phytoplankton), supports the concept that tidal exchange and natural marsh processes are the primary controls on oxygen dynamics in this estuary. In fact, the daily average dissolved oxygen concentration varied inversely with the tidal amplitude suggesting that longer residence time and greater areal submergence of the marsh was responsible for the lowest oxygen observed oxygen levels. Further evidence for the dominance of marsh processes is the lack of linkage between the observed variations in chlorophyll and the extent of oxygen depletion. In embayments, oxygen minima are typically observed as a bloom declines (senesces), a pattern not seen in the Great Marshes (the opposite is recorded by the Scorton Creek mooring) and only slightly in Bass Hole. Consistent with this latter observation is that the long-term average TN level at the mooring sites ranged from 0.43-44 mg N L^{-1} in the central main basin to 0.58-0.74 mg N L^{-1} in the tributary tidal creeks and generally <0.50 mg L^{-1} in the mid and lower waters of Bass Hole, The pattern and magnitude of oxygen depletion again does not follow the observed nitrogen gradients, particularly in Bass Hole, as has been found for most embayment systems. It is clear that the organic rich nature of the upper portions of salt marsh creeks and the low organic deposition in the high velocity areas are a predominant control of oxygen levels. It should be noted that the observed levels of TN is typical of salt marsh creeks, and does not indicate impairment in that type of environment. Overall, based upon the measured levels of oxygen and chlorophyll and comparisons to other unimpaired salt marshes tributary to Cape Cod Bay, it does not appear that the Barnstable Great Marsh estuary system is impaired based on these metrics.

Unlike the bulk of the Barnstable Great Marsh estuary system, the Millway is an artificial basin, dredged for navigation and to support marina activities, similar to Rock Harbor, Orleans. The Millway receives ebbing water from Maraspin Creek, a New England pocket salt marsh. Presently, the Millway functions as an open water basin and as such is more sensitive to nitrogen enrichment than the nearby salt marsh areas. Oxygen levels in the Millway were generally high over the record, however there were periods of significant oxygen depletion for an open water basin, <4 mg L^{-1} for 5% of the record. The main issue is that the low oxygen event, with declines to 2 mg L^{-1}, comprised an event rather than sporadic daily depletions over the 36 day record, which suggests stress to benthic animal communities within this basin. It is likely that a deepened basin within a greater salt marsh system and down gradient from Maraspin Creek Marsh, has created an enhanced depositional environment (for phytoplankton and marsh detritus) where sediment processes may be facilitating oxygen demand and supporting the observed oxygen depletion.

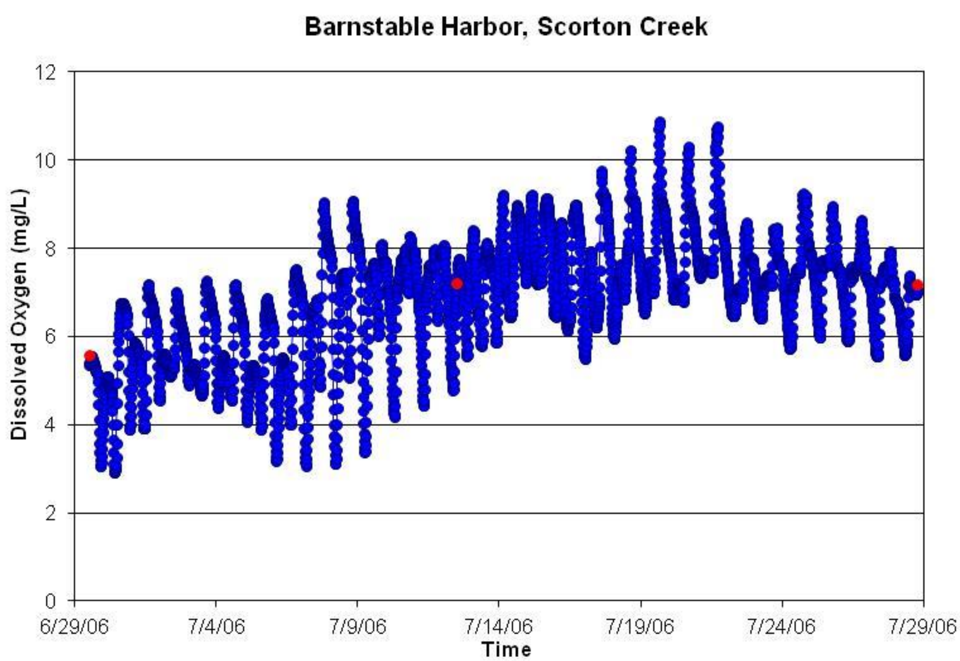


Figure VII-3. Bottom water record of dissolved oxygen at the Barnstable Harbor-Scorton Creek station (BH-01), Summer 2006. Calibration samples represented as red dots.

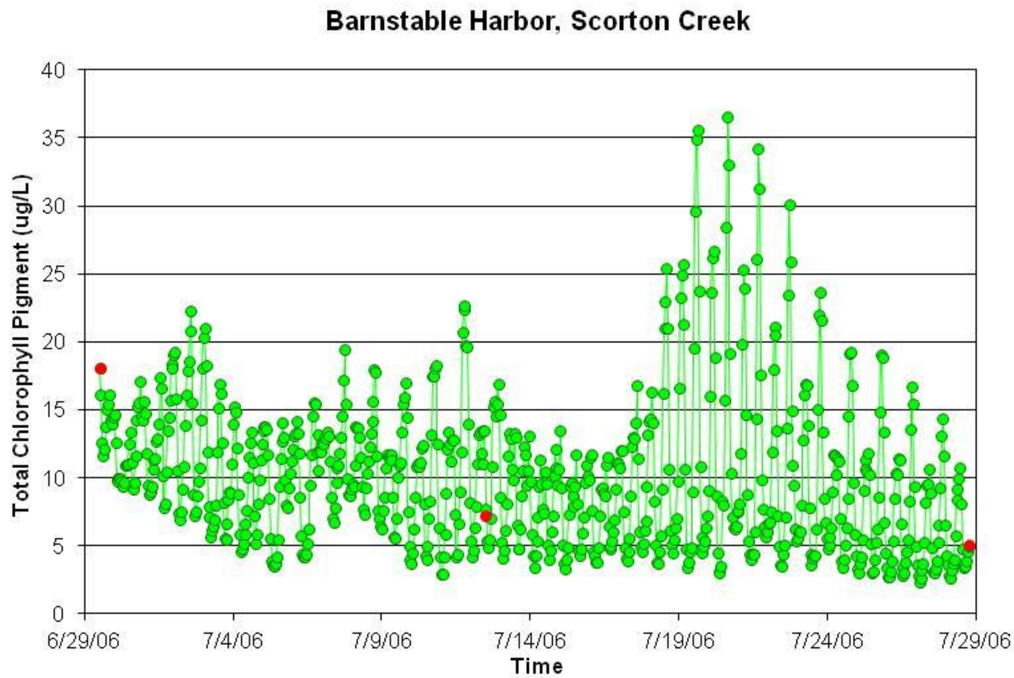


Figure VII-4. Bottom water record of total pigment (Chlorophyll-*a*+pheophytin) in the Barnstable Harbor-Scorton Creek station (BH-01), Summer 2006. Calibration samples represented as red dots.

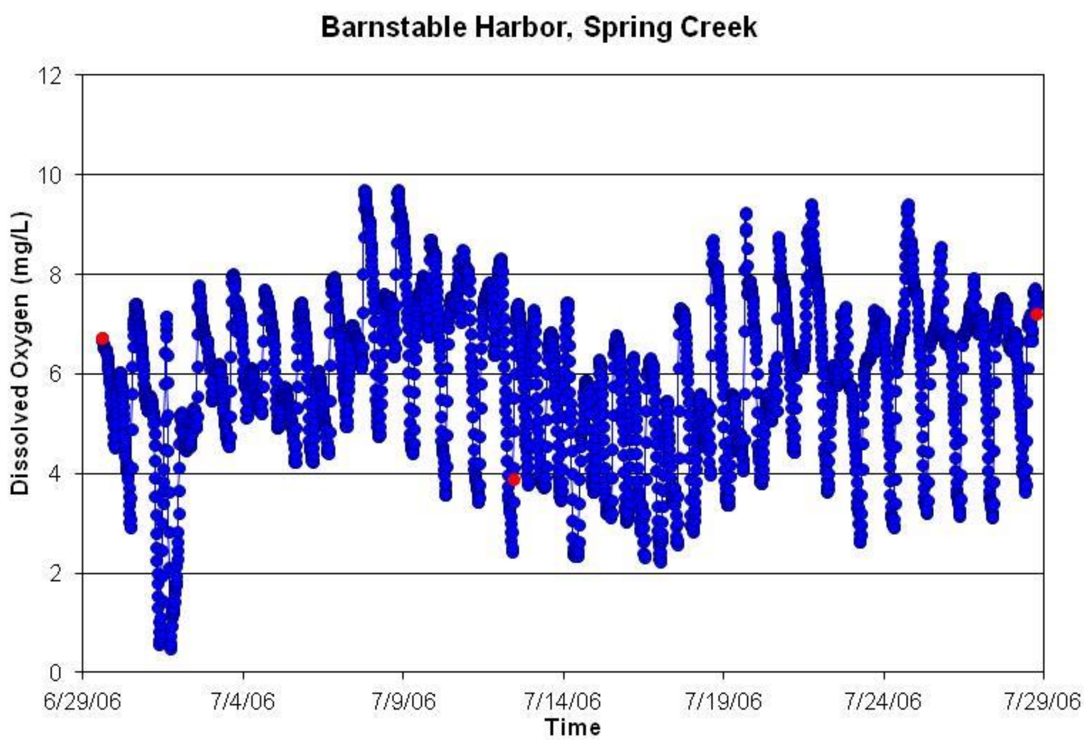


Figure VII-5. Bottom water record of dissolved oxygen at the Barnstable Harbor-Spring Creek station (BH-02), Summer 2006. Calibration samples represented as red dots.

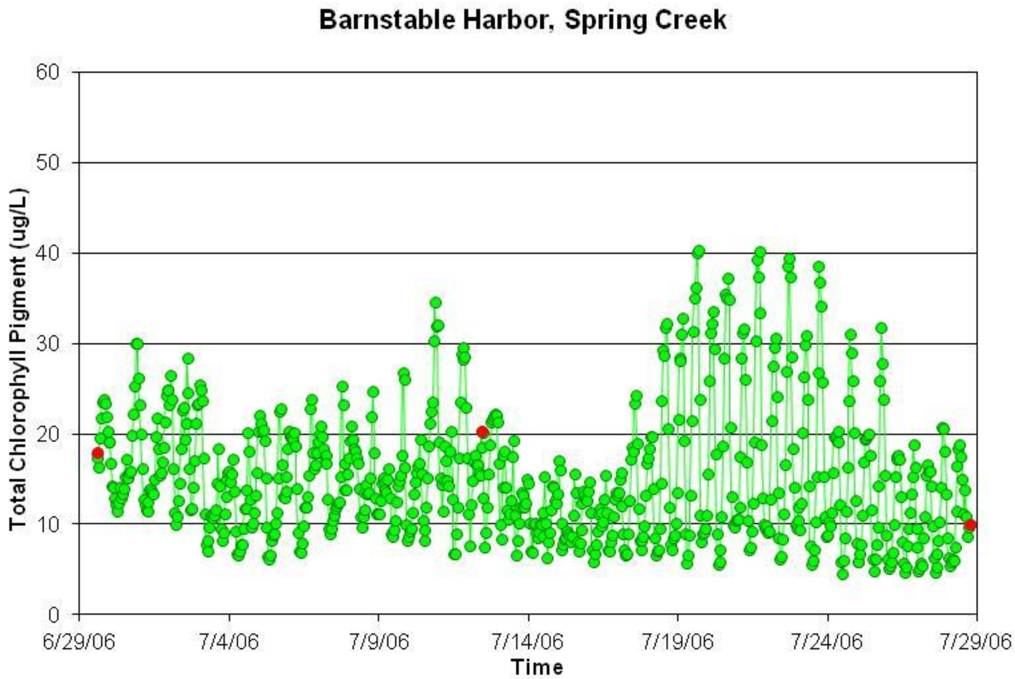


Figure VII-6. Bottom water record of total pigment (Chlorophyll-*a*+pheophytin) in the Barnstable Harbor-Spring Creek station (BH-02), Summer 2006. Calibration samples represented as red dots.

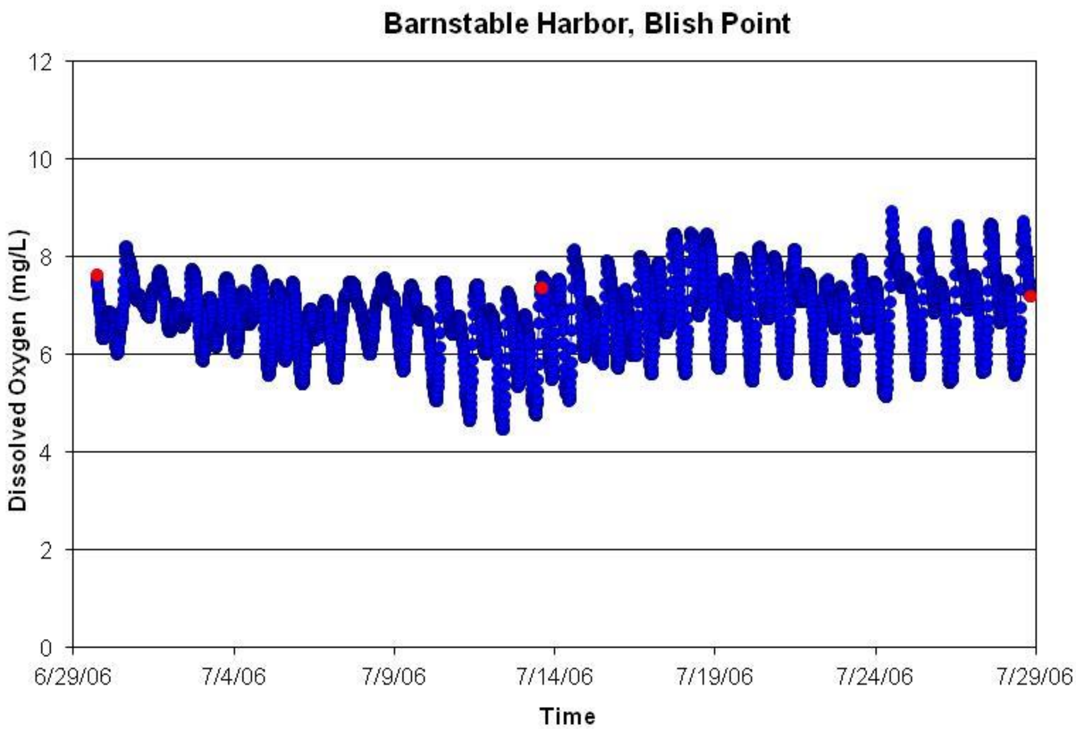


Figure VII-7. Bottom water record of dissolved oxygen at the Barnstable Harbor-Blish Point station (BH-04), Summer 2006. Calibration samples represented as red dots.

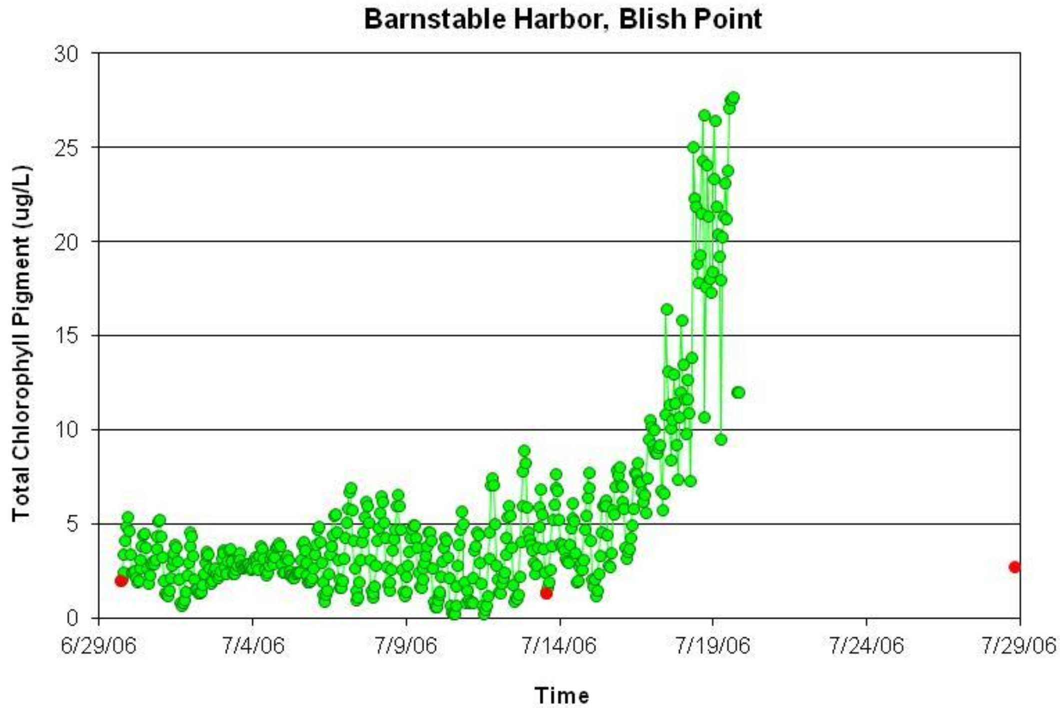


Figure VII-8. Bottom water record of total pigment (Chlorophyll-*a*+pheophytin) in the Barnstable Harbor-Blish Point station (BH-04), Summer 2006. Calibration samples represented as red dots.

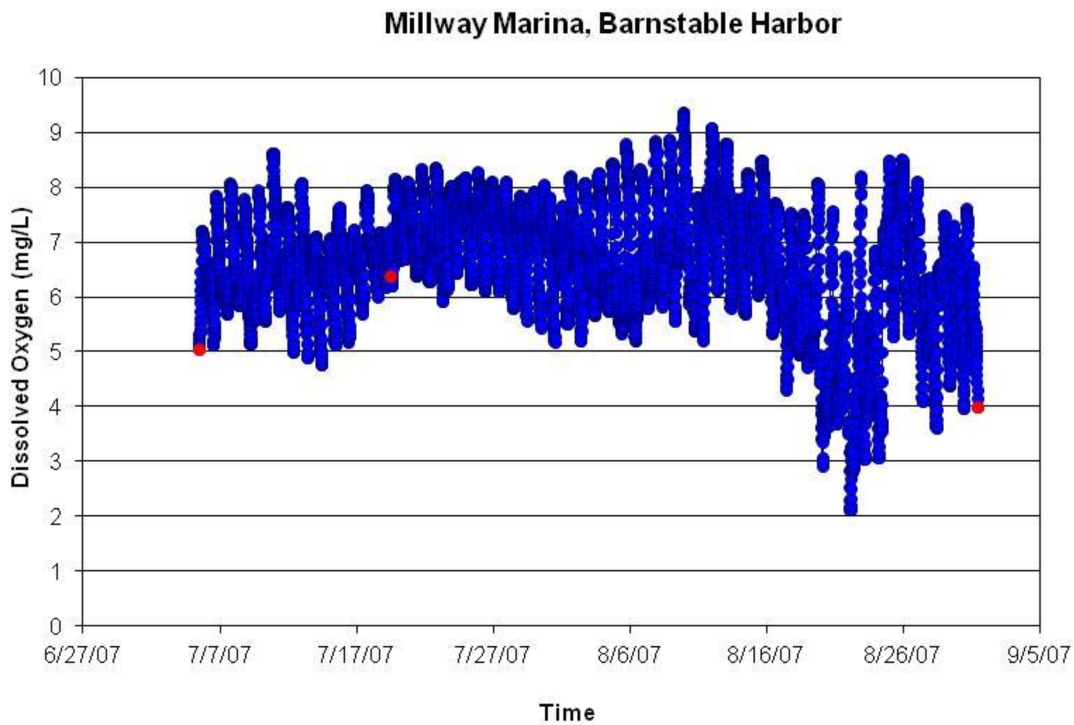


Figure VII-9. Bottom water record of dissolved oxygen at the Barnstable Harbor-Millway Marina station, Summer 2007. Calibration samples represented as red dots.

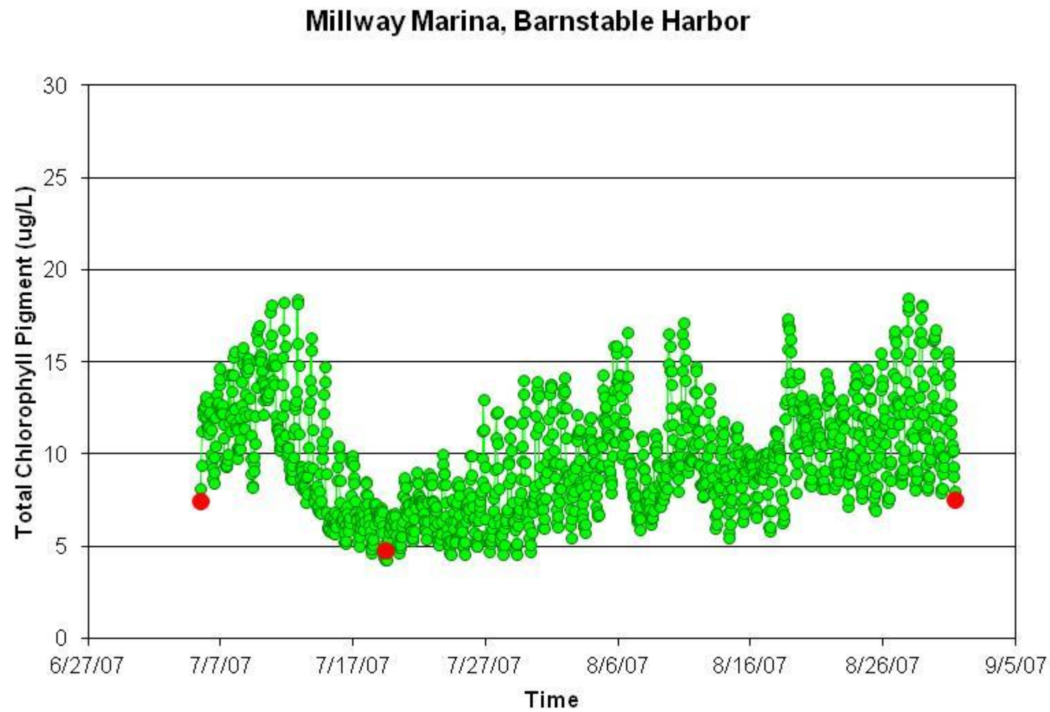


Figure VII-10. Bottom water record of total pigment (Chlorophyll-*a*+pheophytin) in the Barnstable Harbor-Millway Marina station, Summer 2007. Calibration samples represented as red dots.

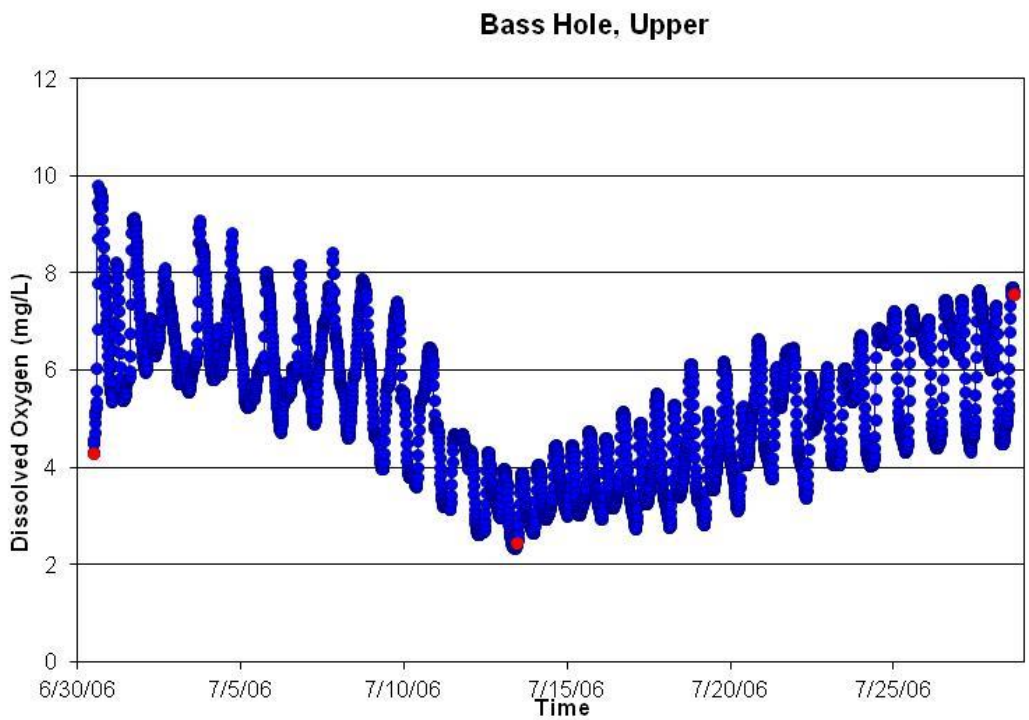


Figure VII-11. Bottom water record of dissolved oxygen at the Bass Hole-Upper station, Summer 2006. Calibration samples represented as red dots.

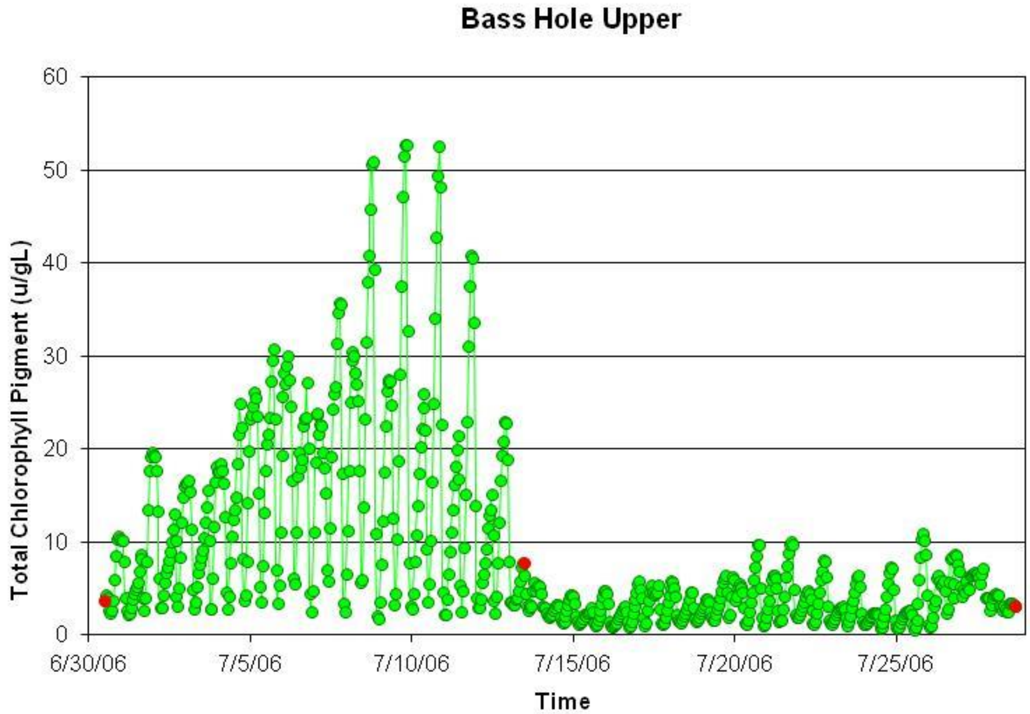


Figure VII-12. Bottom water record of total pigment (Chlorophyll-*a*+pheophytin) at the Bass Hole-Upper station, Summer 2006. Calibration samples represented as red dots.

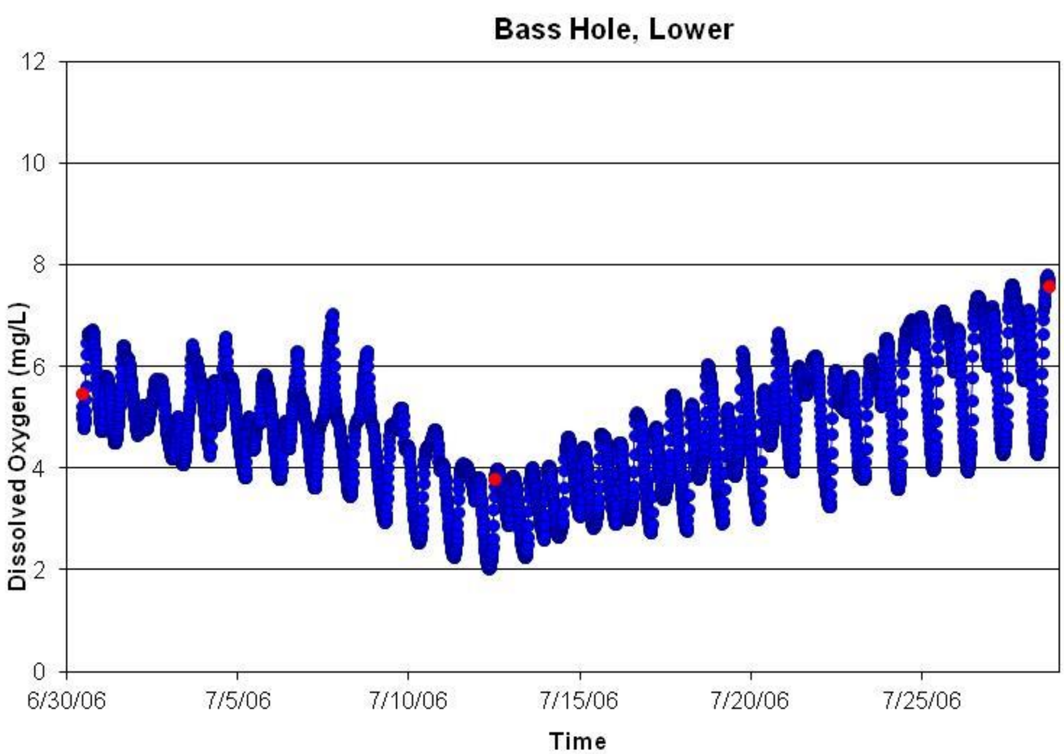


Figure VII-13. Bottom water record of dissolved oxygen at the Bass Hole-Lower station, Summer 2006. Calibration samples represented as red dots.

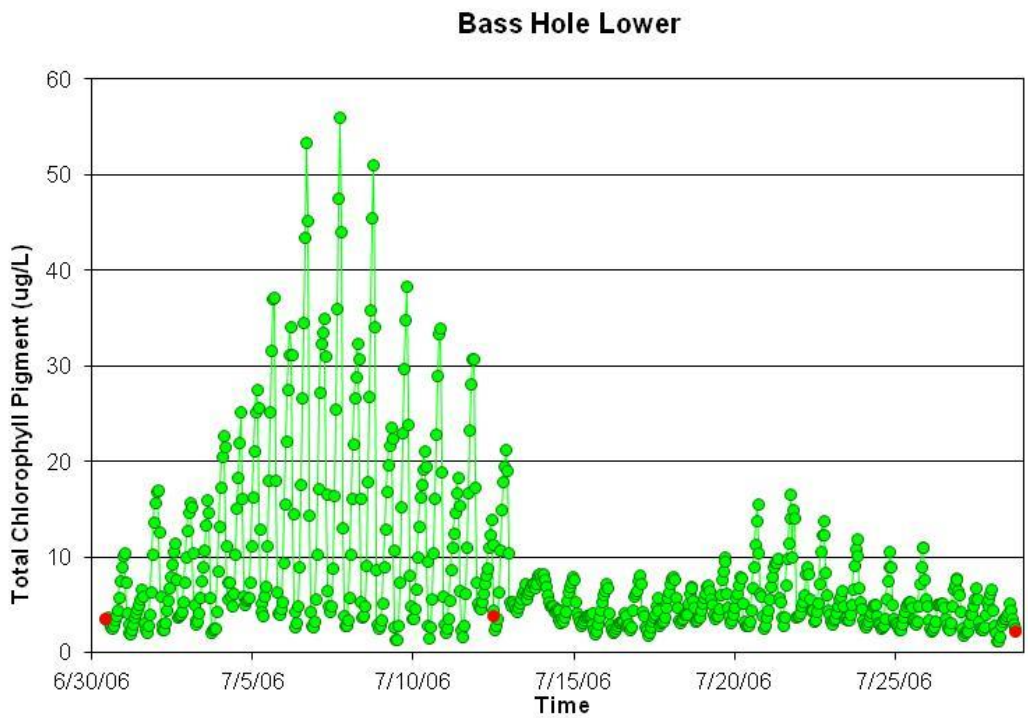


Figure VII-14. Bottom water record of total pigment (Chlorophyll-*a*+pheophytin) in the Bass Hole-Lower station, Summer 2006. Calibration samples represented as red dots.

Table VII-1. Days and percent of time during deployment of in situ sensors that bottom water oxygen was below various benchmark oxygen levels at each of the 6 mooring sites within the Barnstable Great Marsh estuary system. Data collected by the Coastal Systems Program, SMAST.

Mooring Location	Start Date	End Date	Total Deployment (Days)	<6 mg/L Duration (Days)	<5 mg/L Duration (Days)	<4 mg/L Duration (Days)	<3 mg/L Duration (Days)
Bass Hole Lower	**6/30/2006**	**7/28/2006**	**28.3**	**83%**	**60%**	**29%**	**8%**
			Mean	1.12	0.54	0.29	0.15
			Min	0.04	0.06	0.03	0.04
			Max	9.91	6.79	1.88	0.31
			S.D.	2.09	1.18	0.33	0.10
Bass Hole Upper	**6/30/2006**	**7/28/2006**	**28.2**	**66%**	**45%**	**24%**	**5%**
			Mean	0.66	0.56	0.34	0.15
			Min	0.03	0.05	0.01	0.04
			Max	7.86	5.73	1.43	0.32
			S.D.	1.44	1.14	0.28	0.11
Millway Marina	**7/5/2007**	**8/31/2007**	**36.1**	**43%**	**13%**	**5%**	**1%**
			Mean	0.24	0.18	0.09	0.07
			Min	0.01	0.01	0.01	0.01
			Max	0.93	0.77	0.48	0.11
			S.D.	0.22	0.21	0.10	0.04
Scorton Creek	**6/29/2006**	**7/28/2006**	**29.2**	**25%**	**10%**	**3%**	**0%**
			Mean	0.30	0.15	0.09	0.05
			Min	0.03	0.03	0.03	0.05
			Max	1.02	0.31	0.14	0.05
			S.D.	0.29	0.08	0.04	NA
Blish Point	**6/29/2006**	**7/28/2006**	**29.1**	**17%**	**1%**	**0%**	**0%**
			Mean	0.16	0.12	NA	NA
			Min	0.01	0.09	0.00	0.00
			Max	0.28	0.16	0.00	0.00
			S.D.	0.07	0.03	NA	NA
Spring Creek	**6/29/2006**	**7/28/2006**	**29.2**	**49%**	**29%**	**15%**	**4%**
			Mean	0.36	0.20	0.16	0.11
			Min	0.02	0.02	0.01	0.02
			Max	0.94	0.38	0.33	0.30
			S.D.	0.22	0.09	0.08	0.09

Table VII-2. Duration (days and % of deployment time) that chlorophyll-a levels exceed various benchmark levels at each of the 6 mooring sites within the Barnstable Great Marsh estuary system. "Mean" represents the average duration of each event over the benchmark level and "S.D." its standard deviation. Data collected by the Coastal Systems Program, SMAST.

Mooring Location	Start Date	End Date	Total Deployment (Days)	>5 ug/L Duration (Days)	>10 ug/L Duration (Days)	>15 ug/L Duration (Days)	>20 ug/L Duration (Days)	>25 ug/L Duration (Days)
Bass Hole Lower	**6/30/2006**	**7/28/2006**	**28.3**	**54%**	**23%**	**15%**	**9%**	**7%**
Mean Chl Value = 8.5 ug/L			**Mean**	0.30	0.23	0.19	0.16	0.16
			Min	0.04	0.04	0.04	0.04	0.04
			Max	0.96	0.33	0.29	0.21	0.21
			S.D.	0.17	0.08	0.08	0.06	0.05
Bass Hole Upper	**6/30/2006**	**7/28/2006**	**28.3**	**48%**	**28%**	**21%**	**13%**	**8%**
Mean Chl Value = 9.0 ug/L			**Mean**	0.32	0.33	0.28	0.23	0.18
			Min	0.04	0.13	0.04	0.04	0.04
			Max	0.96	0.42	0.38	0.33	0.29
			S.D.	0.22	0.09	0.10	0.08	0.09
Millway Marina	**7/5/2007**	**8/31/2007**	**57.1**	**98%**	**40%**	**5%**	**0%**	**0%**
Mean Chl Value = 9.3 ug/L			**Mean**	4.30	0.44	0.14	NA	NA
			Min	0.04	0.04	0.04	0.00	0.00
			Max	32.08	2.42	0.38	0.00	0.00
			S.D.	9.02	0.37	0.09	NA	NA
Scorton Creek	**6/29/2006**	**7/28/2006**	**29.3**	**81%**	**46%**	**15%**	**5%**	**2%**
Mean Chl Value = 10.2 ug/L			**Mean**	0.63	0.22	0.13	0.12	0.08
			Min	0.04	0.04	0.04	0.08	0.04
			Max	4.75	0.67	0.25	0.17	0.13
			S.D.	0.86	0.12	0.06	0.03	0.04
Blish Point	**6/29/2006**	**7/28/2006**	**20.2**	**30%**	**11%**	**7%**	**4%**	**1%**
Mean Chl Value = 5.2 ug/L			**Mean**	0.34	0.28	0.27	0.18	0.07
			Min	0.04	0.04	0.04	0.08	0.04
			Max	3.42	0.96	0.50	0.38	0.17
			S.D.	0.77	0.31	0.21	0.12	0.06
Spring Creek	**6/29/2006**	**7/28/2006**	**29.4**	**99%**	**73%**	**44%**	**21%**	**10%**
Mean Chl Value = 15.2 ug/L			**Mean**	4.85	0.42	0.23	0.16	0.14
			Min	0.42	0.04	0.04	0.04	0.04
			Max	25.00	2.67	0.63	0.29	0.21
			S.D.	9.87	0.39	0.15	0.09	0.06

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data is key part of the MEP Approach. Surveys were conducted in 1995 and 2001 in the vicinity of the mouth of the Barnstable Great Marsh estuary system by the DEP Eelgrass Mapping Program to be integrated into the MEP effort. These surveys were essentially in the near shore waters of Cape Cod Bay and into the mouth of the Barnstable Harbor system but did not extend into the tidal creeks of the Great Marshes portion of the system or Bass Hole. The primary use of the data is to indicate (a) estuarine regions that have historically or presently support eelgrass habitat, and (b) if large-scale system-wide shifts have occurred. To the extent that surveys have been conducted these data sets can provide a view of temporal trends in eelgrass distribution from 1995 to 2001 to 2010-2013 (Figure VII-15); the period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community. However, MassDEP mapping of the eelgrass beds was not conducted within the component basins of the Barnstable Great Marsh estuary system.

A historical analysis of possible eelgrass distribution (1951) was conducted within the estuary using aerial photos, which showed only a limited area supporting eelgrass. The photographic analysis indicated the possible presence of eelgrass in the open water portion of Barnstable Harbor close to the mouth and east of what is commonly referred to as the Millway. This is likely because many of the central basin areas have very dynamic sediments, with unstable sands that do not support eelgrass. The MEP Technical Team confirmed that eelgrass is not currently present in the tidal creeks to the Great Marshes portion of the system or Bass Hole while undertaking field surveys as part of the benthic regeneration and infauna studies and during the deployment and recovery of the instrument moorings (summer and fall 2007).The absence of any eelgrass, was consistent with a salt marsh dominated system that is also composed of a large open water area with a large tidal range, strong tidal currents and large areas of shifting sand flats and sand waves. As a result of the absence of eelgrass or documentation indicating eelgrass loss due to nitrogen enrichment, temporal changes in eelgrass distribution could not provide a basis for evaluating recent increases (nitrogen loading) in nutrient enrichment of Barnstable Harbor and Bass Hole. It should be noted that the historical eelgrass distribution is not confirmed or validated. None-the-less to the extent that it existed, the cause of its disappearance is consistent with a non-nitrogen factor, due to the low phytoplankton and macroalgal biomass, oxidized sediments, fully oxygenated water column (>6 mg/L in 98% of 133 sampling dates by Barnstable Water Quality Monitoring Program) and high light penetration in that region of the estuary. The most likely cause appears to be associated with high tidal velocities, unstable sediments and possibly winter storm exposure. Unstable sediments (shifting sands) have been identified as the mechanism causing eelgrass loss adjacent the 2007 tidal inlet to Pleasant Bay, and similarly the very poor habitat for benthic animals in portions of Chatham Harbor. Aerial photos of the Great Marshes document the extensive swept sands (Figure VII-2, VII-15,16) in the eastern region of the Great Marshes open water basin and tidal inlet. In addition, the absence of eelgrass from the tidal creeks like those tributary to the Great Marsh is typical of Cape Cod salt marshes which do not generally support eelgrass habitat, particularly when the creeks nearly completely drain during each ebb tide The absence of eelgrass, for at least the past 60 years, has been documented for Namskaket and Little Namskaket Marshes, to the north also on Cape Cod Bay as well as Sandwich Harbor and Scorton Creek to the west of Barnstable Harbor, all salt marsh dominated systems with large tidal ranges on Cape Cod Bay.

In contrast to the estuarine basins, an area of eelgrass is present offshore slightly to the east of the inlet to Barnstable Harbor, based on the 1995, 2001 and 2010-2013 eelgrass survey conducted by the DEP Eelgrass Mapping Program. However, there was no evidence of the eelgrass bed in the same area during the 2006 survey of this small offshore bed. It is not possible at this time to determine if this represents an anthropogenically driven decline or natural variation at this site, however, given the dynamic nature of this area, it is possible that the lack of eelgrass during the 2006 survey could be the result of storm activity and vigorous littoral transport. Additional spatial and temporal sampling undertaken by the MassDEP Eelgrass Mapping Program in the future might be a way to better understand shifts in eelgrass beds in that area.

Based on the salt marsh dominated function of large portions of the Barnstable Great Marsh estuary system, historical absence of eelgrass in all of the other salt marsh systems on Cape Cod that exchange waters with Cape Cod Bay and the similarity of these tidal creeks to Sandwich Harbor, Scorton Creek, Namskaket Creek and Little Namskaket Creek that have also not historically supported eelgrass habitat, the MEP Technical Team concludes that the Barnstable Great Marsh estuary system has not supported eelgrass for many decades, if not longer. Based upon all available information, it appears that the Barnstable Great Marsh estuary system is not structured to support eelgrass habitat. Therefore, threshold development for protection/restoration of this system will focus on infaunal animal habitat quality. This is typical for New England salt marsh dominated estuaries, which are naturally organic and nutrient rich and generally contain little water in the creeks at low tide as well as having open water areas with strong tidal currents and actively shifting sand bottoms near their tidal inlets. This conclusion has been confirmed in a wide range of salt marsh dominated basins throughout southeastern Massachusetts by the MEP Technical Team.

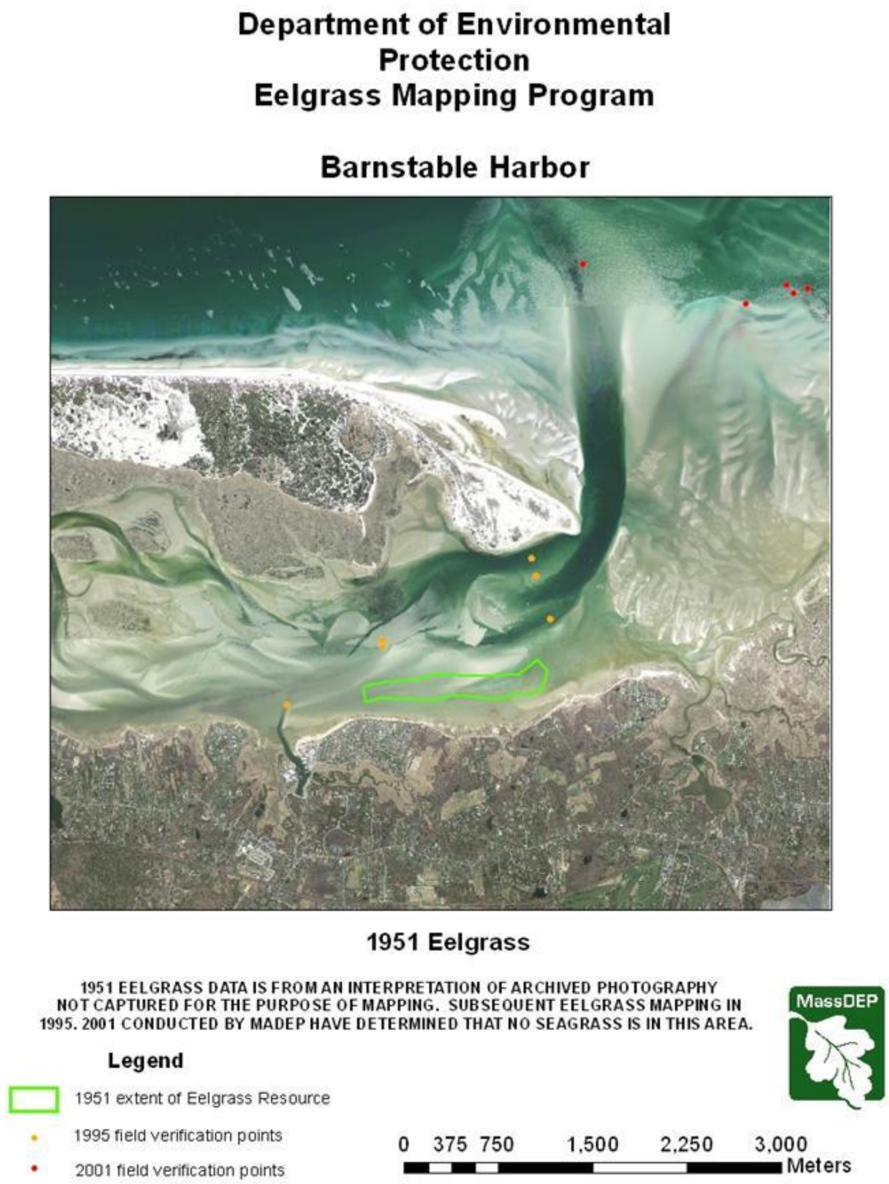


Figure VII-15. Estimated eelgrass bed distribution in Barnstable Harbor based on photo-interpretation of 1951 aerial. Beds were not identified in 1995 and 2001 (map from the MassDEP Eelgrass Mapping Program). MassDEP surveying did not extend into the salt marsh tidal creeks, however, no eelgrass was observed in the Barnstable Great Marsh estuary system during SMAST-MEP surveying in 2007.

Figure VII-16. Eelgrass bed distribution offshore of the Barnstable Great Marsh estuary system (2010-2013). Beds were not identified in 1995 and 2001 (map from the MassDEP Eelgrass Mapping Program). MassDEP surveying did not extend into the salt marsh tidal creeks, however, no eelgrass was observed in any of the component basins during SMAST-MEP surveying in 2007.

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 23 locations within the major tidal creeks and open water areas throughout the Barnstable Great Marsh estuary system (Figure VII-17), with replicate assays at each site. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity (H') and Evenness (E) of the community. It should be noted that, although there are no eelgrass beds in the entirety of the system, this is almost certainly due to the system functioning as a large intertidal salt marsh as well as the fact that the open water area is dynamic and dominated by shifting sand flats. The Great Marshes and Bass Hole portions of the system, like all the other large salt marshes tributary to Cape Cod Bay, do not appear to have historically supported eelgrass habitat and therefore the absence of eelgrass is "natural" and does not indicate impairment. As such, to the extent that the overall embayment system can support healthy infaunal communities given specific nutrient conditions in the water column, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Section VIII).

Barnstable Harbor Infaunal Characteristics:

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information (Table VII-3). The evenness statistic can range from 0-1 (one being most even), while the diversity index does not have a theoretical upper limit. The highest quality habitat areas, as shown by the oxygen and chlorophyll records and eelgrass coverage, have the highest diversity (generally >3) and evenness (~0.7). The converse is also true, with poorest habitat quality found where diversity is <1 and evenness is <0.5. The infauna study of the salt marsh dominated portions of the estuary and the open water lower main basin indicated that the tidal creeks are presently supporting a salt marsh infaunal habitat typical of large unimpaired salt marsh systems on Cape Cod with open basins in their lower reaches. Infauna communities within the tidal creeks were indicative of the fine grained organic rich environment typical of salt marshes or of high velocity areas with sandy sediments and were consistent with the observed levels of oxygen depletion and water column TN. The communities within the upper reach had high numbers of individuals with moderate numbers of species and diversity and Evenness as also was found in the tributary creeks in the lower estuary, while the lower open basin supported highly productive communities with moderate to high diversity and Evenness. Only the small artificial marina basin of the Millway showed any impairment of benthic habitat. All of the areas surveyed supported almost no stress indicator organisms (Tubificids, Capitellids) and were generally dominated by polychaetes (Spionids) with some crustaceans and mollusks. Most species were deposit feeders, similar to other New England salt marshes. Areas of high tidal water velocities in some portions of the main channel have winnowed the fines from the sediments and created a medium to coarse sand with shifting sediments. The result is similar to tidal inlets or Chatham Harbor's shifting sands which do not support extensive benthic habitat, being naturally disturbed. The observed communities in the tributary creeks and upper main basin were typical of New England salt marsh creek bottom environments in summer. The soft-bottomed areas supported communities with organic enrichment tolerant species and were dominated by *Streblospio* and polychaetes, the major invertebrate family comprising the benthos. Also present were significant numbers of crustaceans and mollusks. Sediments throughout the estuary (exception of Millway) were oxidized and consolidated with no observed anoxia, but periodic oxygen depletion typical of salt marshes, high water quality and low to moderate chlorophyll-*a.* In addition, the bioactive nitrogen levels (dissolved inorganic N = particulate organic N, i.e. DIN + PON) were generally low (long term averages, <0.16 mg N L^{-1}) except in upper creeks at low tide where creek water is highly modified by groundwater inflows. Bioactive N was selected as the critical nitrogen form for analysis of the Barnstable Great Marsh estuary system due to the very high background dissolved organic nitrogen (DON) associated with the extensive salt marsh areas. Bioactive nitrogen was also the critical management element in the Pleasant Bay Estuary MEP analysis for the same reasons (see Howes et al. 2006 and Chapter VIII-2). This approach is supported by very low reactivity of dissolved organic matter in coastal waters, with measurements age of over 1,000 years. Moreover, while the DIN in bioactive N is readily available for plant uptake on the order of minutes to hours and PON can settle to the sediments and be degraded, the refractory nature of DON gives is a negligible role in this estuary's potential eutrophication. Given the high rate of water turnover within the Barnstable Great Marsh estuary system would mean that the nitrogen in DON would need to be biologically available within 24-48 hours, the flushing time of most of the main basin and tidal creeks comprising this system or it is removed to Cape Cod Bay without having an effect on the estuary.

Overall, the infauna survey indicated that all areas within the Barnstable Great Marsh estuary system are supporting healthy infauna habitat typical of open salt marsh dominated basins or organic rich New England salt marsh tidal creeks, hence high quality relative to each specific estuarine ecosystem type. The mid-lower main channel is partially structured by its areas of high water velocity and physical disturbance to the sediment. The lack of nitrogen related impairment is supported by the general absence of surface algal mats and macroalgae, with only a few patches of very sparse *Ulva* or *Gracillaria* being observed. The near absence of macroalgal accumulations is consistent with the relatively low bioactive nitrogen levels within the saline waters of this system, <0.13 mg N L^{-1} (long-term average). By comparison, high quality benthic habitat in sub-basins of the system was found at 0.21 mg L^{-1}. Similarly in the adjacent Scorton Creek Salt Marsh, high quality habitat was found in areas with bioactive N <0.18 mg/L and nearby Sandwich Harbor Estuary with extensive marshes had similarly high quality habitat with similar bioactive N levels $\leq$0.16 mg/L., All of these salt marsh dominated systems had similar environmental conditions, with macroalgal accumulations nearly absent. Based upon all lines of evidence it appears that the Great Marshes and Bass Hole basins are presently supporting high quality infaunal habitat and have not exceeded their threshold nitrogen levels for assimilating additional nitrogen without impairment. The exception is the Millway which functions as a depositional basin functioning as an embayment tributary to the Great Marshes. It appears that the deep basin to support marina and navigation activities is a focal point for deposition of phytoplankton and marsh detritus. The result is benthic habitat consisting of soft unconsolidated (fluid) organic rich sediments without an oxidized surface layer that are sulfidic. The result is a depauperate benthic animal community with <10 organisms per grab, more than an order of magnitude less than even impaired habitats. At present the Millway supports a severely degraded benthic habitat, with very low productivity.

Table VII-3. Benthic infaunal community data for the Barnstable Great Marsh estuary system, which is a major tidal salt marsh system tributary to Harbor. Measured number of species and individuals, with estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations. Samples represent surface area of 0.0625 m^2. Stations refer to map in Figure VII-10.

Sub-Embayment	Station ID	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Barnstable Great Marshes						
Central Main Basin – lower	BH20,22,26,27	16	680	12	2.53	0.64
Central Main Basin – upper	BH 9,15,17	12	137	7	2.32	0.67
Scorton Creek	BH 1,3,4	8	254	7	1.50	0.51
Spring Creek	BH 6,7	5	193	5	1.40	0.69
Brickyard Creek	BH10,11,12,14	9	819	7	1.77	0.62
Millway/Barnstable Hbr	BH 24	1	9	na	0.32	0.32
Wharf Creek	BH 30	17	300	13	2.74	0.68
Bass Hole						
Bass Hole – lower	Bass 1-3	19	147	14	2.47	0.58
Bass Hole – upper	Bass 5-6	21	412	13	2.90	0.66

Figure VII-17. Aerial photograph of the Barnstable Great Marsh estuary system showing location of benthic infaunal sampling stations (green symbols).

Other Resource Characteristics:

In addition to benthic infaunal community characterization undertaken as part of the MEP field data collection, other biological resources assessments were integrated into the habitat assessment portion of the MEP nutrient threshold development process as developed by the Commonwealth and available. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest (Figure VII-18a,b,c) as well as the suitability of a system for the propagation of shellfish (Figure VII-19). As is the case with few systems on Cape Cod, the majority of Barnstable Great Marsh estuary system is classified as approved for the taking of shellfish at any time during the year. In a few nearshore areas the system is conditionally approved, generally the tidal creeks leaving salt marshes or the areas in the vicinity of discharging tidal creeks. The conditional approval classification for these areas during specific times of the year is most likely due to bacterial inputs from wildlife and birds associated with the wetland areas of the system. One area commonly referred to as the Millway is classified as prohibited to shellfishing as an active marina and possibly due to inputs from storm water, septic systems, and commercial activity. In conjunction with existing shellfish area classifications, the Barnstable Great Marsh estuary system is also classified as supportive of specific shellfish communities (Figure VII-12). The major shellfish species with potential habitat within the Barnstable Great Marsh estuary system are soft shell clams (*Mya*) in the more open water areas that are dominated by sandy bottom and quahogs (*Mercenaria*) that appear to dominate the upper and lower portions of the tidal channel network. Interspersed among the areas suitable for soft shell clams and quahogs are small areas suitable for surf clams.

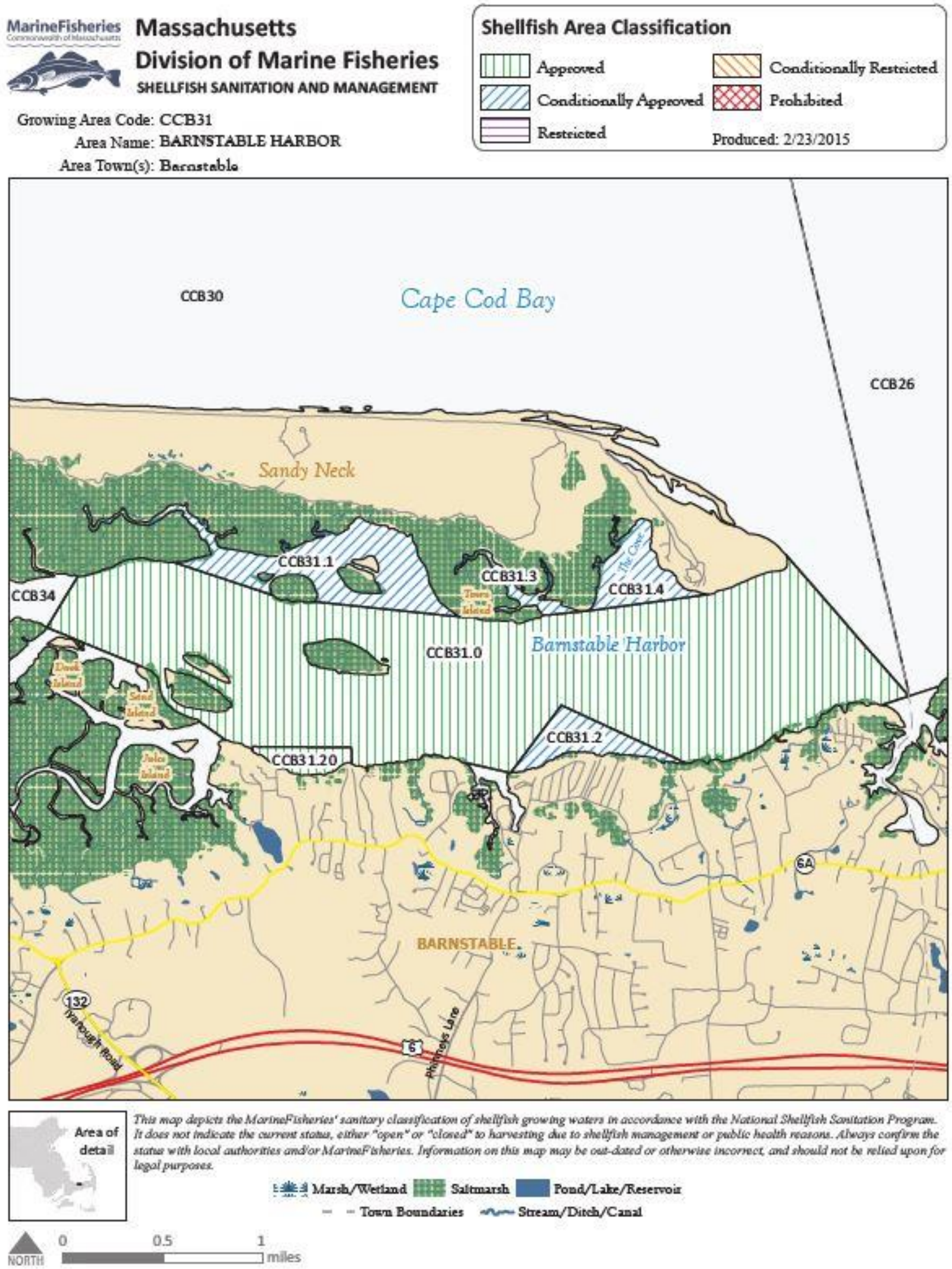


Figure VII-18a. Location of shellfish growing areas in the Barnstable Harbor embayment system and the status relative to shellfish harvesting as determined by Massachusetts Division of Marine Fisheries. Closures are generally related to bacterial contamination from wildlife or human "activities", such as the location of marinas, septic tanks or stormwater discharges.

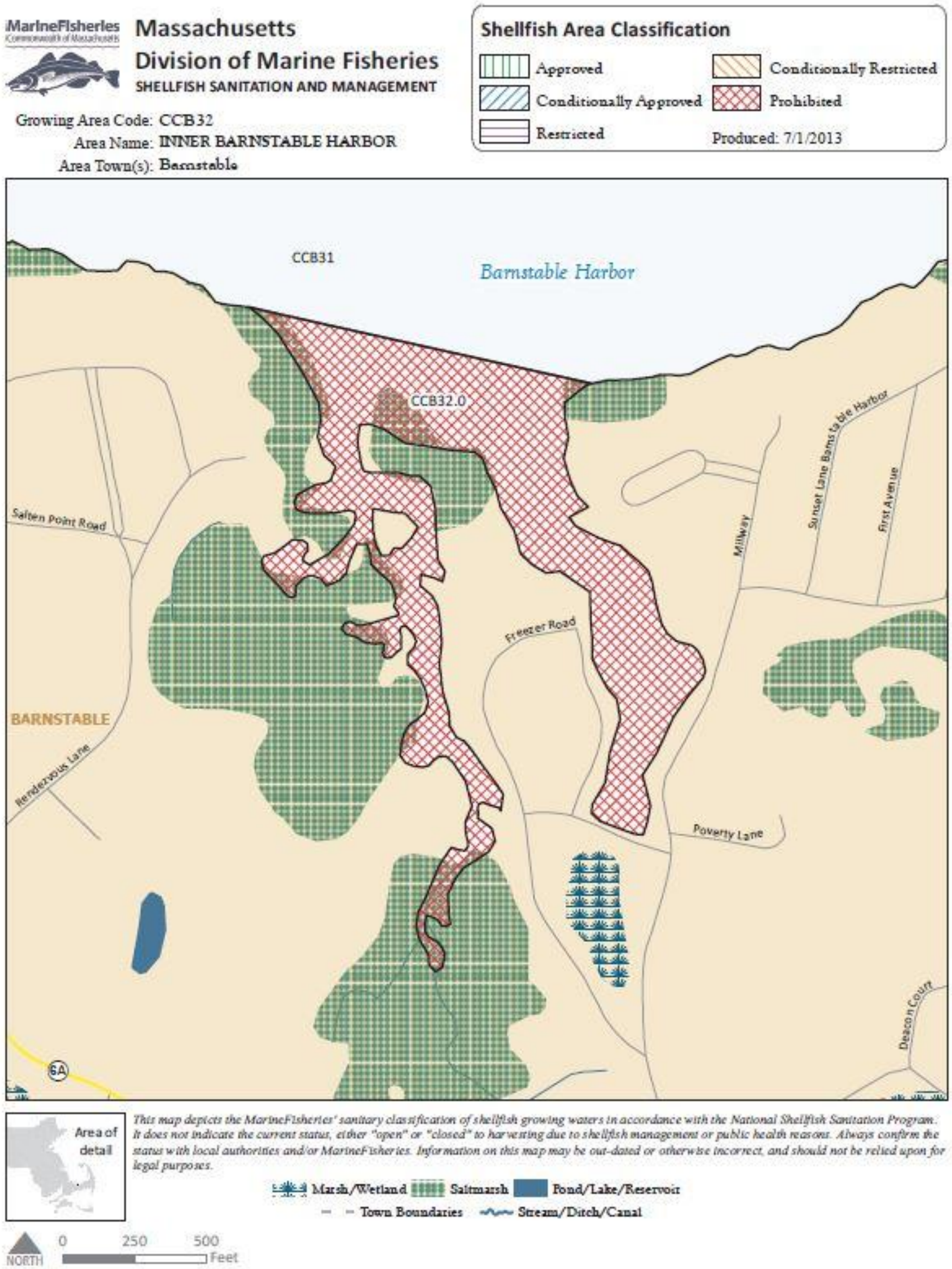


Figure VII-18b. Location of shellfish growing areas in the Millway portion of the Barnstable Harbor embayment system and the status relative to shellfish harvesting as determined by Massachusetts Division of Marine Fisheries. Closures are generally related to bacterial contamination from wildlife or human "activities", such as the location of marinas, septic tanks or stormwater discharges.

Figure VII-18c. Location of shellfish growing areas in the Great Marshes portion of the Barnstable Harbor embayment system and the status relative to shellfish harvesting as determined by Massachusetts Division of Marine Fisheries. Closures are generally related to bacterial contamination from wildlife or human "activities", such as the location of marinas, septic tanks or stormwater discharges.

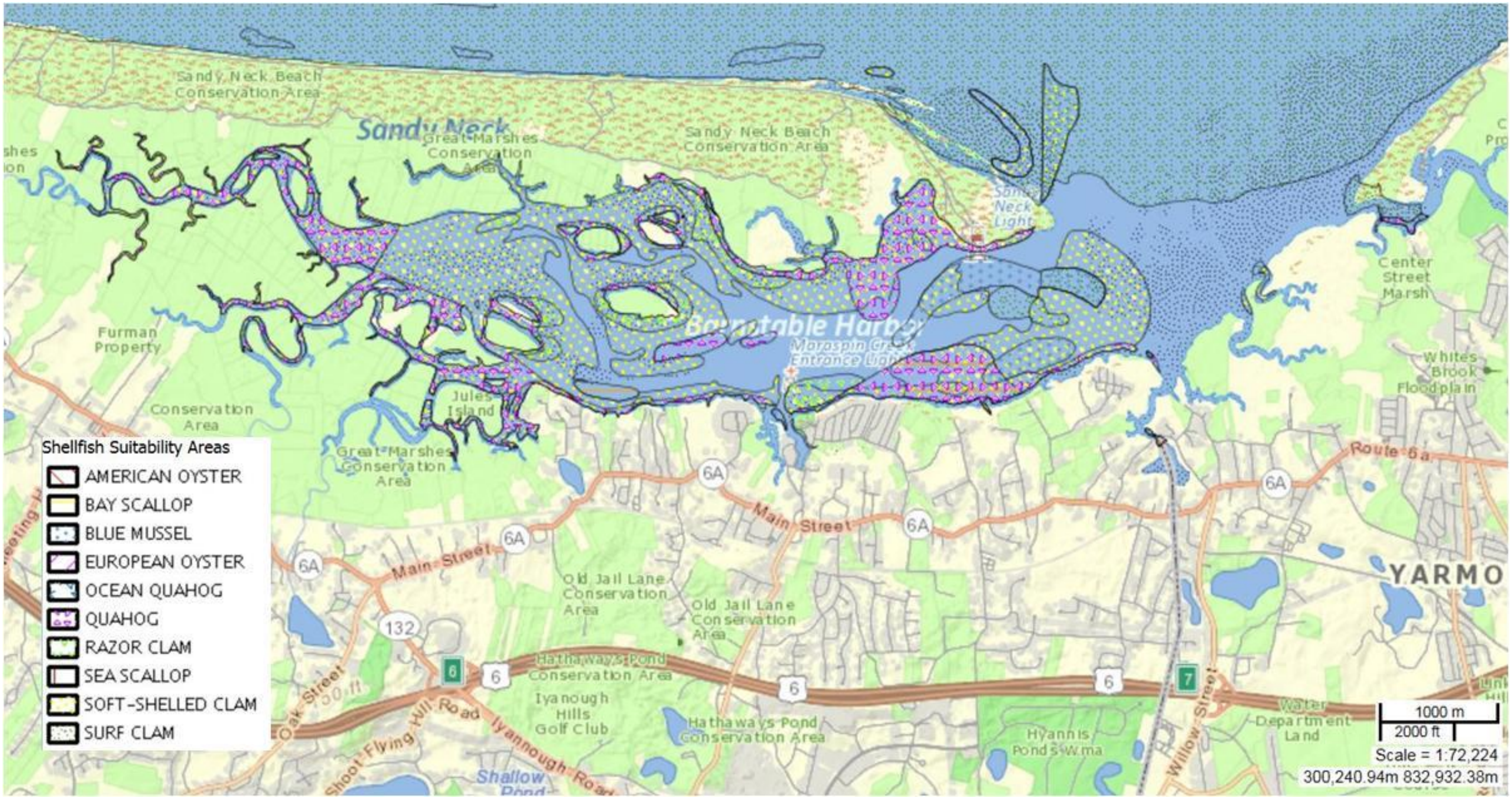


Figure VII-19. Location of shellfish suitability areas within the Barnstable Harbor estuary as determined by Massachusetts Division of Marine Fisheries. Suitability does not necessarily mean "presence". The delineated areas generally coincide with creek bottoms dominated by fine and medium sand.

VIII. CRITICAL NUTRIENT THRESHOLD DETERMINATION AND DEVELOPMENT OF WATER QUALITY TARGETS

VIII.1. ASSESSMENT OF NITROGEN-RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires the integration of key habitat parameters (infauna and eelgrass), sediment characteristic data, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll a). Additional information on temporal changes within each sub-embayment and its watershed further strengthen the analysis. These data were all collected to support threshold development within the component sub-embayments comprising the Barnstable Great Marsh estuary system by the MEP Team and were discussed in Section VII. Nitrogen threshold development builds on these data and links habitat quality to summer water column nitrogen levels from long-term baseline water quality monitoring (Towns of Barnstable and Dennis Water Quality Monitoring Programs, and MEP Technical Team).

The Barnstable Great Marsh estuary system is a complex estuary composed of 3 types of basins: 1) tidal embayments (open water basins with little associated salt marsh), the Millway; 2) open water lower basin with tributary salt marsh creeks (salt marsh dominated open basins), main basin of the Great Marshes; and 3) salt marsh tidal creeks some with organic rich sediments within extensive salt marsh which contain little water at low tide and some with high velocities and areas of shifting sands. Each of these 3 basins has a different natural sensitivity to nitrogen enrichment and organic matter loading and each has its own benthic community indicative of an unimpaired or impaired habitat. Evaluation of habitat quality considered the natural structure of each system and the types of infaunal communities that they support under low and high levels of nitrogen enrichment. Infaunal habitat is the focus of protection/restoration in this complex estuary as eelgrass has not been documented and presumptive eelgrass loss to the extent that it is real, does not appear to be associated with nutrient enrichment At present, the Barnstable Great Marsh estuary system is showing differences in nitrogen enrichment and habitat quality among its various component basins (Table VIII-1).

In field surveys since the 1990's, the Barnstable Great Marsh estuary system have not supported eelgrass habitat. Based on the salt marsh dominated function of large portions of the Barnstable Great Marsh estuary system, historical absence of eelgrass in all of the other salt marsh systems on Cape Cod that exchange waters with Cape Cod Bay and the similarity of these tidal creeks to Sandwich Harbor, Scorton Creek, Namskaket Creek and Little Namskaket Creek, that have also not historically supported eelgrass habitat, the MEP Technical Team concludes that the Barnstable Great Marsh estuary system has not supported eelgrass for many decades, if not longer. Based upon all available information, it appears that the Barnstable Great Marsh estuary system is not structured to support eelgrass habitat. Therefore, threshold development for protection/restoration of this system will focus on infaunal animal habitat quality. This is typical for New England salt marsh dominated estuaries, which are naturally organic and nutrient rich and generally contain little water in the creeks at low tide as well as having open water areas with strong tidal currents and actively shifting sand bottoms near their tidal inlets. This conclusion has been confirmed in a wide range of salt marsh dominated basins throughout southeastern Massachusetts by the MEP Technical Team.

Portions of the estuary and the open water lower main basin indicate that the tidal creeks are supporting a salt marsh infaunal habitat typical of large unimpaired salt marsh systems on

Cape Cod with open basins in their lower reaches. Infauna communities within the tidal creeks were indicative of the fine grained organic rich environment typical of salt marshes or in some cases of high velocity channel areas with sandy sediments. Communities were consistent with the observed levels of oxygen depletion, phytoplankton biomass, the lack of macroalgal accumulations and water column nitrogen The communities within the upper reaches of the creeks had high numbers of individuals with moderate numbers of species and diversity and Evenness as was also found in the tributary creeks in the lower estuary. The lower open basin supported highly productive communities with moderate to high diversity and evenness. Only the small artificial marina basin of the Millway showed any impairment of benthic habitat. All of the areas surveyed supported almost no stress indicator organisms (Tubificids, Capitellids) and were generally dominated by polychaetes (Spionids) with some crustaceans and mollusks. Most species were deposit feeders, similar to other New England salt marshes.

The observed communities in the tributary creeks and upper main basin were typical of New England salt marsh creek bottom environments in summer. The soft-bottomed areas supported communities with organic enrichment tolerant species and were dominated by *Streblospio* and polychaetes were the major invertebrate family comprising the benthos, but with also significant numbers of crustaceans and mollusks. Sediments throughout the estuary were oxidized and consolidated with no observed anoxia, but showed periodic oxygen depletion typical of salt marshes, high water quality and low to moderate chlorophyll-*a*.

Given the strong salt marsh influences on these basins, which tend to reduce species numbers and diversity even in "pristine" systems, it appears that the salt marsh dominated sub-basins are not showing indications of excessive nutrient enrichment and are currently supporting high quality habitat. It should be noted that salt marshes are naturally nutrient and organic matter enriched and the benthic animal communities found within these basins are consistent with salt marsh influenced systems throughout the region. Similarly, the main creeks (tidal channels) do not appear to be impaired by nitrogen enrichment, although they have fewer species and numbers than open water bays. The creeks do generally support only few stress indicator species The larger tidal channels (usually near the mouth) appear to have shifting sediments due to the very high tidal velocities. Sand waves are common as are shifting sand bars. The MEP has encountered similar conditions in other high velocity channels, with the similar finding of a reduced benthic community, composed of non-organic stress indicators. In these regions the community appears to be structured primarily by the unstable sediments, rather than effects associated with water column conditions (nitrogen, oxygen, chlorophyll).

The exception to the high quality benthic habitat is the Millway which is a depositional basin functioning as an embayment tributary to the Great Marshes. It appears that the deep basin to support marina and navigation activities is a focal point for deposition of phytoplankton and marsh detritus. The result is benthic habitat consisting of soft unconsolidated (fluid) organic rich sediments that are sulfidic and without an oxidized surface layer. The result is a depauperate benthic animal community with <10 organisms per grab sample, more than an order of magnitude less than even impaired habitats. At present the Millway supports a severely degraded benthic habitat, with very low productivity which will persist as long as it remains a depositional site or until the organic matter available for settling is reduced (i.e. phytoplankton).

The results of the evaluations of the key habitat indicators (infaunal animals, eelgrass. dissolved oxygen/chlorophyll-*a*) coupled with macroalgal survey data were used to assess the overall habitat quality of each component sub-embayment to the Barnstable Great Marsh estuary system (Table VIII-1). The results of the habitat assessment show consistent assessments

Table VIII-1. Summary of Nutrient Related Habitat Health within the Barnstable Great Marsh estuary system, a salt marsh dominated estuary on Cape Cod, MA., based upon assessment data presented in Chapter VII. D.O. (dissolved oxygen) and Chl a (chlorophyll a) from the mooring data (VII.2). WQMP=Town Water Quality Monitoring Program results.

Sub-Embayment	Nutrient related Health Indicator					
	D.O.	**Chl a**	**Macro-algae**	**Eelgrass**	**Infaunal Animals**	**Overall**
Barnstable Great Marshes						
Central Main Basin -- lower	H[2]	H[8]	--[13]	--[14,15]	H[16]	**H[19]**
Central Main Basin – upper	H[2]	H[8]	--[13]	--[14]	H[17]	**H[20]**
Scorton Creek	H[1,3]	H[9]	--[13]	--[14]	H[17]	**H[20]**
Spring Creek	H[1.3]	H[9]	--[13]	--[14]	H[17]	**H[20]**
Brickyard Creek	H[1,4]	H[10]	--[13]	--[14]	H[17]	**H[20]**
Millway/Barnstable Hbr	MI/SI[5]	MI[12]	--[13]	--[14]	SD[18]	**SD[18, 21]**
Wharf Creek	--[6]	--[6]	--[13]	--[14]	H[16]	**H[20]**
Bass Hole (Chase Garden Creek)						
Bass Hole – lower	H[1,7]	H[11]	--[13]	--[14]	H[16]	**H[20]**
Bass Hole – upper	H[1,7]	H[11]	--[13]	--[14]	H[16]	**H[20]**

1) natural oxygen depletions, typical of salt marsh creeks, which can go anoxic at night due to natural high organic sediments and high oxygen uptake..
2) generally >6 mg/L infrequent oxygen depletions to 5-6 mg/L, 3%-5% of WQMP samples, mid basin mooring (Blish Pt) >6 mg/L 82% of record, rarely 5mg/L, 1% of record. .
3) tidal creeks almost always >4 mg/L infrequent oxygen depletions to 4-5 mg/L, 3%of WQMP; >4 mg/L at Scorton and Spring Creeks, 97% & 85% of record, rarely 3 mg/L.
4) highly organic marsh creek: lower reach DO to 3 mg/L 4% of WQMP, uppermost reaches frequently 2-3 mg/L, 30%-46% of samples, never <2mg/L as in many salt marshes
5) oxygen depletion, >5 mg/L 87% of mooring record, but periodic events to <4mg/L 5% of record, unconsolidated sediments highly reducing (sulfidic) and organic enriched.
6) no data, but sediments highly oxidized
7) oxygen depletion typical of saltmarsh creeks, frequently to 3 mg/L, ~5% of mooring record; WQMP >4mg/l 98% of dates lower basin, frequently 3 mg/L in upper small creeks,
8) low summer chlorophyll levels averaging 2.8-4.5 ug/L on 133 WQMP dates, 5.2 ug/L at mid basin mooring with bloom to 20 ug/L.
9) low-moderate chlorophyll levels 7.3-7.4 ug/L on 35 WQMP dates, bloom to 20 ug/L with means of 10-15 ug/L over mooring record in these main tidal creeks.
10) low-moderate = 7-8 ug/L (lower) and 10-14 ug/L (smaller upper creeks), WQMP 26 dates
11) low-moderate chlorophyll levels <4 ug/L (lower) and ~12 ug/L (upper) creek on 45 WQMP dates, mooring record similar to WQMP, but for late bloom to 20 ug/L with means of ~9 ug/L upper & lower
12) moderate chlorophyll for embayment mooring average 9.3 ug/L, bloom to >15 ug/L
13) drift algae sparse or absent
14) no evidence this basin is supportive of eelgrass.
15) no evidence presumptive eelgrass loss due to nitrogen enrichment
16) high # species (16) & individuals (>250) or moderate individuals (~150), high-moderate diversity (.2.5) & evenness (0~0.65),dominated by polychaetes with mollusks and crustaceans with few stress indicator species.
17) for saltmarshes: typical # species & individuals, moderate diversity & evenness, dominated by polychaetes (Spionids) with variable crustaceans with few stress indicator species.
18) severely depleted benthic community, <10 individuals per grab, consistent with the observed soft "fluid" sulfidic sediments
19) marsh open basin: high DO, low Chla,, macroalgae absent with productive benthic community
20) typical high quality saltmarsh, naturally organic/nutrient enriched with moderate DO and Chla levels, supporting typical benthic communities, with few stress indicator species.
21) as embayment: moderate DO depletion and Chla levels, but degraded benthic habitat

H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment; SD = Severe Degradation; -- = not applicable to this estuarine reach between indicators and follow the long-term levels of water column nitrogen (see Section VIII.3, below). All of these data were integrated in the development of the nitrogen thresholds for the protection of infaunal habitats throughout the Barnstable Great Marsh estuary system and restoration of these habitats within the Millway (Section VIII.2).

VIII.2. THRESHOLD NITROGEN CONCENTRATIONS

The approach for determining nitrogen loading rates, which will maintain acceptable habitat quality throughout and embayment system, is to first identify a sentinel location within the embayment and second to determine the nitrogen concentration within the water column which will restore that location to the desired habitat quality (threshold nitrogen level). The sentinel location is selected such that the restoration of that one site will necessarily bring the other regions of the system to acceptable habitat quality levels. Once the sentinel site and its target nitrogen level are determined, the Linked Watershed-Embayment Model is used to sequentially adjust nitrogen loads until the targeted nitrogen concentration is achieved. For the Barnstable Great Marsh estuary system, the protection/restoration target should reflect both recent pre-degradation habitat quality and be reasonably achievable.

The threshold nitrogen level for an embayment represents the tidally averaged watercolumn concentration of nitrogen that will support the habitat quality being sought. The watercolumn nitrogen level is ultimately controlled by the watershed nitrogen load and the nitrogen concentration in the inflowing tidal waters (boundary condition). The watercolumn nitrogen concentration is modified by the extent of sediment regeneration.

The threshold nitrogen level for the Barnstable Great Marsh estuary system was developed to restore or maintain high habitat quality. High habitat quality in this case is defined as supportive of healthy and productive infaunal communities. Dissolved oxygen and chlorophyll-*a* were considered in the assessment as was macroalgae occurrence and distribution. Due to the complexity of the Barnstable Great Marsh estuary system and distances between sub-basins and complex circulation, three “sentinel” stations were developed targeting the Great Marshes, the Millway and Bass Hole, as these 3 component basins are functioning somewhat independently. The Great Marshes sentinel station is based upon the average of adjacent upper marsh stations BH-11 and BH-12, long-term water quality monitoring stations, while the Bass Hole station BSH-4 is at the upper boundary of the main tidal creek. The Millway station was developed (as for the Three Bays Estuary) using the modeled observed concentrations. The thresholds use tidally average nitrogen levels from the water quality model of the long-term measurements. The MEP uses tidally averaged nitrogen concentrations as it increases the accuracy, by reflecting what the environment experienced over complete tidal cycles rather than at a single point in the cycle.

The approach developed by the MEP has been to select sentinel stations based upon location within a system, generally in the upper tidal reaches as this is typically where water quality is lowest. Therefore, restoration or protection of the sentinel sub-embayment will necessarily create high quality habitat throughout the estuary. Second, the sentinel station should be in a sufficiently large basin to prevent steep horizontal water quality gradients, such as would be found in the region of entry of a stream or river or in the upper most region of a narrow, shallow estuary. This second criteria relates to the ability to accurately determine the baseline nitrogen level and to conduct the predictive modeling runs. Finally, the site of the sentinel station should be able to obtain the minimum level of habitat quality acceptable for the greater system (unless a multiple classification is to be used).

After the sentinel sub-system (or systems) is selected, the nitrogen level associated with high and stable habitat quality typically derived from a lower reach of the same system or an adjacent estuary or estuaries is used to develop the nitrogen concentration target. Finally, the watershed nitrogen loading rate is manipulated in the calibrated water quality model to determine the watershed nitrogen load which will produce the tidally averaged target nitrogen level at the sentinel location (Section VIII-3 below). Differences between the required modeled nitrogen load to achieve the target nitrogen level and the present watershed nitrogen load represent nitrogen management goals for restoration or protection of the embayment system as a whole.

Based upon the absence of eelgrass throughout the Barnstable Great Marsh estuary system (Chapter VII), infaunal animal habitat was selected as the target for the development of the site-specific nitrogen threshold.

The MEP's analysis of the Barnstable Great Marsh estuary system found very high levels of dissolved organic nitrogen within the embayment's waters (based upon data from the Barnstable and Dennis Water Quality Monitoring Programs). While some small portion of the dissolved organic nitrogen is actively cycling, the vast majority is refractory (non-biologically active) within the timeframe of the flushing of the overall system. The result is that the dissolved organic nitrogen presents a large non-active pool generally separate from the nitrogen fractions active in eutrophication and impacts to habitat quality (i.e. ammonium and nitrate+nitrite, particulate organic nitrogen). The biologically active nitrogen pools are represented by the species directly available to phytoplankton and algae (plant available nitrogen), ammonium and nitrate+nitrite, and the particulate organic nitrogen comprised primarily of phytoplankton (live and dead). Together this nitrogen group is termed bioactive nitrogen. Given the large dissolved organic nitrogen pool within Barnstable Great Marsh estuary system, the MEP Technical Team adopted the same approach used previously for the TMDL analysis of Pleasant Bay. In this previous analysis, the threshold was developed based upon the bioactive nitrogen pool, which appears to be relatively consistent between embayments both within and outside of Pleasant Bay as it is for the Barnstable Great Marsh estuary system. Equally important the dissolved organic nitrogen component in TN, but not part of bioactive nitrogen, is relatively uniform (unchanging) throughout the Barnstable Great Marsh estuary system averaging 0.228 mg L^{-1} across all stations (2.03 x BactN + 0.228 = TN mg/L, R^2=0.8), indicating its low reactivity.

In meeting the threshold value and achieving restoration, the bioactive nitrogen threshold has less uncertainty than the total nitrogen threshold given the biogeochemistry of these systems. Therefore, while both values form the basis for guiding nitrogen reductions to achieve ecological restoration, the total nitrogen value should only be evaluated in light of the bioactive nitrogen threshold. Critical nitrogen threshold levels were developed to support healthy infaunal animal habitat, see below.

The level of bioactive nitrogen supportive of high quality infaunal community habitat appears to be relatively constant among salt marsh dominated basins. Therefore, the MEP Technical Team set a single threshold of 0.16 mg L^{-1} for these basins, and 0.21 mg L^{-1} for the sub-embayment of the Millway, 0.21 mg L^{-1} has been used in Bassing Harbor and Pleasant Bay to restore high quality infaunal communities in the associated salt ponds and coves not supportive of eelgrass. Given the similar structure and function of the Millway, this was deemed to be justified and a reasonable approach. The lower levels for the salt marsh dominated sites reflects that they are currently well below the 0.16 threshold and are not showing impairment.

The infaunal habitat threshold was derived in a similar manner to the site-specific eelgrass threshold relying depends heavily upon the present distribution of infaunal communities relative to water column nitrogen levels and measured oxygen depletions. For embayment basins, like the Millway, data from moderately impaired infaunal communities in Ryders Cove with tidally averaged bioactive nitrogen levels of 0.244 mgN L^{-1}; moderately impaired infaunal communities, in The River adjacent the inlet to Lonnies Pond (0.217 mgN L^{-1}), the Namequoit River (0.216-0.239 mgN L^{-1}). While healthy infaunal habitat in the lower Pochet Basin is found at bioactive N levels of 0.18 mgN L^{-1}. For these basins, it appears that the infaunal threshold lies between 0.18 and 0.22 mgN L^{-1} tidally averaged bioactive nitrogen. Based upon the summary above, animal community and nitrogen analysis in enclosed embayment basins tributary to larger estuaries indicates that restoration/protection of a healthy habitat would target a bioactive nitrogen threshold of 0.21 mgN L^{-1},

In the tributary salt marsh creeks and salt marsh dominated basins, the tidally averaged bioactive nitrogen levels were generally low (~0.11 mg L^{-1}) except in the upper-most reaches of the creeks at low tide where creek water is highly modified by groundwater inflows. Bioactive N was selected as the critical nitrogen form for analysis of the Barnstable Great Marsh estuary system infauna habitat threshold, due to the very high background dissolved organic nitrogen (DON) associated with the extensive salt marsh areas.

At present, the infaunal animal habitat in all areas within the Barnstable Great Marsh estuary system are supporting high quality infauna communities, typical of open salt marsh dominated basins or organic rich New England salt marsh tidal creeks, hence high quality relative to each specific estuarine ecosystem type. The mid-lower main channel is partially structured by its areas of high water velocity and physical disturbance to the sediment. The lack of nitrogen related impairment is supported by the general absence of surface algal mats and macroalgae, with only a few patches of very sparse *Ulva* or *Gracillaria* being observed. The near absence of macroalgal accumulations is consistent with the relatively low bioactive nitrogen levels within the saline waters of this system, <0.13 mg N L^{-1} (long-term average) By comparison, high quality benthic habitat in Pleasant Bay sub-basins was found at 0.21 mg L^{-1}. Similarly in adjacent Scorton Creek Salt Marsh high quality habitat was found in areas with bioactive N <0.18 mg/L and nearby Sandwich Harbor Estuary with extensive marshes had similarly high quality habitat with similar bioactive N levels $\leq$0.16 mg/L., All of these salt marsh dominated systems had similar environmental conditions, with macroalgal accumulations nearly absent. Based upon all lines of evidence it appears that the Great Marshes and Bass Hole basins are presently supporting high quality infaunal habitat and have not exceeded their threshold nitrogen levels for assimilating additional nitrogen without impairment. It also appears that the tidally averaged threshold bioactive nitrogen level for these systems that can be supported from the comparative assessments is 0.16 mg L^{-1}. The nitrogen loads associated with the threshold concentration at the sentinel locations are discussed in Section VIII.3, below.

VIII.3 DEVELOPMENT OF TARGET NITROGEN LOADS

The tidally averaged total nitrogen thresholds derived in Section VIII-2 were used to adjust the calibrated constituent transport model developed in Section VI. The nitrogen thresholds were used to determine the amount of total nitrogen mass loading reduction required for restoration of or to prevent impairments to infaunal habitats in the Barnstable Great Marsh estuary system. Contrary to most estuarine systems evaluated as part MEP, the threshold concentration was set higher than present conditions, meaning that the system would be allowed to have a higher load than present and still be able to meet the threshold. Other MEP studies that resulted in an allowable increased threshold loading have all had extensive salt marsh resources (like Sandwich Harbor and Namskaket Marsh), which is the case for both Barnstable Great Marshes and Bass Hole.

Therefore, for Barnstable Great Marshes, watershed nitrogen loads were sequentially raised in the model until the nitrogen levels reached the tidally averaged bioactive N concentration at the selected sentinel stations (BM-11, BM-13 and BSH-4) selected for the threshold (0.16 mg/L). One new site was added in the Barnstable Harbor/Millway, which did not have a long-term monitoring station. For the Millway, loads where required to be lowered in order to achieve the 0.21 mg/L threshold concentration set for this small basin. The Millway is a dredged basin with upgradient salt marsh that is depositional and supports degraded sediment infaunal communities. A similar basin and ecological response was detailed for the Rock Harbor Estuary (Orleans), which is dredged and depositional and also supports impaired benthic animal communities, although the up-gradient salt marsh creek is unimpaired.

It is important to note that load changes to achieve the nitrogen management threshold could be produced by changing of any or all sources of nitrogen to the system. The load changes presented below represent only one of a suite of potential approaches that could to be evaluated by the Town. The threshold scenario presented here is used to establish the general degree and spatial pattern of loading that will be allowable for this system. A comparison between present septic and total watershed loading and the loadings for the modeled threshold scenario is provided in Tables VIII-1 and VIII-2, respectively.

As shown in Table VIII-2, the threshold set for this system would allow up to a 39.6% increase the total watershed loading. In this particular scenario run, the watershed increase in achieved solely by increasing the total septic load to the system by 53.1% (increasing the total septic load by about one-half). The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

Table VIII-3 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. In Table VIII-3, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent ‘worst-case’ summertime conditions. The benthic flux for this modeling effort is modified from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Cape Cod Bay, as discussed in Section VI.

The comparison between model results of existing loading conditions and the selected loading scenario to achieve the target TN concentrations is shown in Table VIII-4. To achieve the allowable ensemble threshold nitrogen concentration, increases in average TN concentrations of typically greater than 30% occur in Bass Hole (Chase Garden Creek) and more than 45% in the western end of the Great Marshes (Table VIII-5).

Although the above modeling results provide one manner of achieving the selected threshold level for the system, the specific example does not represent the only method for achieving this goal. However, this example provides a general sense of what could be possible when considering future N loading increases to the Harbor.

Table VIII-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading scenarios of the Barnstable Great Marsh estuary system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.			
sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Barnstable Great Marshes – west	26.364	50.737	+92.4%
Barnstable Great Marshes – mid	19.488	48.719	+150.0%
Barnstable Great Marshes - east	32.397	32.397	0.0%
Millway	7.205	2.522	-65.0%
Bass Hole – west	23.107	30.385	+31.5%
Bass Hole – east	20.822	36.438	+75.0%
Bass Hole	5.847	5.847	0.0%
System Total	135.230	207.045	+53.1%

Table VIII-3. "Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios for the Barnstable Great Marsh estuary system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loads. Note that the Great Marshes are also called "Barnstable Harbor" and Bass Hole, "Chase Garden Creek by residents.			
sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Barnstable Great Marshes - west	38.586	62.959	+63.2%
Barnstable Great Marshes - mid	27.562	56.793	+106.1%
Barnstable Great Marshes - east	42.600	42.600	+0.0%
Millway	10.575	5.892	-44.3%
Bass Hole – west	29.666	36.944	+24.5%
Bass Hole – east	25.030	40.647	+62.4%
Bass Hole	7.408	7.408	0.0%
System Total	181.427	253.242	+39.6%

Table VIII-4. Threshold sub-embayment loads used for total nitrogen modeling of the Barnstable Harbor system, with total watershed N loads, atmospheric N loads, and benthic flux. Note that the Great Marshes are also called “Barnstable Harbor” and Bass Hole, “Chase Garden Creek by residents.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Barnstable Great Marshes - west	62.959	2.422	6.758
Barnstable Great Marshes - mid	56.793	13.800	-12.351
Barnstable Great Marshes - east	42.600	32.978	-22.405
Millway	5.892	0.211	2.350
Bass Hole – west	36.944	0.140	0.408
Bass Hole – east	40.647	0.003	-
Bass Hole	7.408	0.488	-1.190
System Total	253.242	50.041	-26.430

Table VIII-5. Comparison of model average bioactive N concentrations from present loading and the threshold scenario, with percent change over background in Cape Cod Bay (0.063 mg/L), for the Barnstable Harbor system.

Station Location	monitoring station (MEP ID)	present (mg/L)	threshold (mg/L)	% change
Scorton Creek	BM-13	0.121	0.156	+60.0%
Spring Creek	BM-11	0.127	0.161	+55.0%
Great Marshes – upper	BM-12	0.111	0.139	+57.4%
Great Marshes – upper	BM-1	0.097	0.116	+56.9%
Great Marshes – mid	BM-2	0.085	0.096	+48.0%
Great Marshes – lower	BM-3	0.072	0.077	+51.1%
Broad Sound	BM-10	0.105	0.126	+47.4%
Bass Hole – Inlet	BSH-1	0.093	0.104	+36.8%
Bass Hole – lower	BSH-2	0.111	0.128	+35.8%
Bass Hole – mid	BSH-3	0.122	0.144	+37.2%
Bass Hole – mid	BSH-4	0.132	0.158	+37.2%
Bass Hole – upper	BSH-5	0.271	0.366	+45.5%
Bass Hole – upper	BSH-6	0.571	0.695	+24.4%
Whites Brook	BSH-7	0.321	0.361	+15.3%

Figure VIII-1. Contour plot of tidally averaged modeled total nitrogen concentrations (mg/L) in the Barnstable Great Marsh estuary system at the nitrogen threshold loading. Yellow markers indicate sentinel stations used to determine the threshold (average of BM-11, BM-13, BSH-4 and the Millway station).

IX. LIST OF REFERENCES

AFCEE (with Howes, B.L. & Jacobs Engineering). 2000. Ashumet Pond Trophic Health Technical Memorandum. AFCEE/MMR Installation Restoration Program, AFC-J23-35S18402-M17-0005, 210pp.

Alexander, R.B., J.K. Bohlke, E.W. Boyer, M.B. David, J.W. Harvey, P.J. Mulholland, S.P. Seitzinger, C.R. Tobias, C. Tonitto, W.M. Wollheim. 2009. Dynamic modeling of nitrogen losses in river networks unravels the coupled effects of hydrological and biogeochemical processes. *Biogeochemistry* (2009) 93:91–116. DOI 10.1007/s10533-008-9274-8.

Anders, F.J., and M.R. Byrnes. 1991 Accuracy of Shoreline Change Rates as Determined from Maps and Aerial Photographs. Shore and Beach 16:17-26.

Aravena, R. and W.D. Robertson. 1998. Use of Multiple Isotope Tracers to Evaluate Denitrification in Ground Water: Study of Nitrate from a Large-Flux Septic System Plume. Ground Water, 36(6):975-982.

Brawley, J.W., G. Collins, J.N. Kremer, C.H. Sham, and I. Valiela, 2000. A time-dependent model of nitrogen loading to estuaries form coastal watersheds. Journal of Environmental Quality 29:1448-1461.

Brigham Young University, 1998. “User’s Manual, Surfacewater Modeling System.”

Burns, K., M. Ehrhardt, B. Howes and C. Taylor., 1993. Subtidal benthic community respiration and production rates near the heavily oiled coast of Saudi Arabia. Marine Pollution Bull. 27:199-205.

Cambareri, T.C. and E.M. Eichner, 1998. Watershed Delineation and Ground Water Discharge to a Coastal Embayment. *Ground Water*; 36(4): 626-634.

Cambareri, T.C. and E.M. Eichner, 1998. Watershed Delineation and Ground Water Discharge to a Coastal Embayment. *Ground Water*. 36(4): 626-634.

Cape Cod Commission Water Resources Office, 1991. Technical Bulletin 91-001, Nitrogen Loading.

Cape Cod Commission Water Resources Office, 1998. Cape Cod Coastal Embayment Project Interim Final Report.

Cape Cod Commission, 1998. Cape Cod Coastal Embayment Project: A Nitrogen Loading Analysis of Popponesset Bay. Cape Cod Commission Technical Report.

Cape Cod Commission, 1998. “Cape Cod Coastal Embayment Project.” Barnstable, MA.

Cape Cod Commission. 1996. Regional Policy Plan. Cape Cod Commission, Barnstable, MA.

Cape Cod Commission. 2001. Regional Policy Plan. Cape Cod Commission, Barnstable, MA.

CDM and Howes, B.L., 2000 Water Quality Evaluation of the Wareham River Estuary Complex. Final Report to the Town of Wareham. Camp, Dresser and McKee Inc. 300pp.

Chow, V. T. (1959). Open Channel Hydraulics, McGraw-Hill, NY.

Cod Pond and Lake Atlas. Cape Cod Commission. Barnstable, MA. 275 pp.

Conover, J.T., 1958. Seasonal Growth of Benthic Marine Plants as Related to Environmental Factors in an Estuary. *Institute of Marine Science*, 5:97-147.

Cooksey, C. J. Harvey, L. Harwell, J. Hyland, and J. Summers. 2010. Ecological condition of coastal ocean and estuarine waters of the south Atlantic Bight: 2000-2004. NOAA Technical Memorandum NOS NCCOS 114, NOAA National Ocean Service, Charleston, S.C. 29412-9110; and EPA/600/R-10/046, U.S. EPA, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Gulf Ecology Division, Gulf Breeze FL, 32561, 88 pp.

Costa, J.E., B.L. Howes, I. Valiela and A.E. Giblin. 1992. Monitoring nitrogen and indicators of nitrogen loading to support management action in Buzzards Bay. In: McKenzie et al. (eds.) Ecological Indicators Chapter. 6, pp. 497-529.

Costa, J.E., G. Heufelder, S. Foss, N.P. Millham, B.L. Howes, 2002. Nitrogen Removal Efficiencies of Three Alternative Septic System Technologies and a Conventional Septic System. Environment Cape Cod 5(1): 15-24.

Costello, Charles. Section Chief, Wetlands Conservancy Program. Director, Eelgrass Mapping Program 617-292-5907.

Crowell, M., S.P. Leatherman, M.K. Buckley. 1991. Historical Shoreline Change: Error Analysis and Mapping Accuracy. Journal of Coastal Research 7(3):839-852.

D'Elia, C.F, P.A. Steudler and N. Corwin. 1977. Determination of total nitrogen in aqueous samples using persulfate digestion. Limnology and Oceanography 22:760-764.

DeMoranville, C., Howes, B., Schlezinger, D. and White, D. 2009. Cranberry Phosphorous Management: How Changes in Practice can Reduce Output in Drainage Water. *Acta Hort.* (ISHS) 810:633-640

DeSimone, L.A. and B.L. Howes. 1996. Denitrification and nitrogen transport in a coastal aquifer receiving wastewater discharge. *Environmental Science and Technology* 30:1152-1162.

Dyer, K.R., 1997. Estuaries, A Physical Introduction, 2nd Edition, John Wiley & Sons, NY, 195 pp.

Eichner, E. 2008. Barnstable Ponds: Current Status, Available Data, and Recommendations for Future Activities. School of Marine Science and Technology, University of Massachusetts Dartmouth and Cape Cod Commission. New Bedford and Barnstable, MA. 79 pp.

Eichner, E. 2008. Lake Wequaquet Water Quality Assessment. Completed for the Town of Barnstable and the Cape Cod Commission. Coastal Systems Program, School of Marine Science and Technology, University of Massachusetts Dartmouth. 81 p.

Eichner, E. 2009. Lake Wequaquet Water Quality Assessment. Completed for the Town of Barnstable and the Cape Cod Commission. Coastal Systems Program, School of Marine Science and Technology, University of Massachusetts Dartmouth. 81 pp.

Eichner, E. and B. Howes. March, 2011. Lake Wequaquet 2010 water quality monitoring. Coastal Systems Program Technical Memorandum to Rob Gatewood, Director, Conservation Division, Town of Barnstable. Coastal Systems Program, School of Marine Science and Technology, University of Massachusetts Dartmouth. 11 pp.

Eichner, E., B. Howes, and D. Schlezinger. February, 2015. Lake Wequaquet Dissolved Oxygen Monitoring Program: August – October 2014. Coastal Systems Program Technical Memorandum to Rob Gatewood, Director, Conservation Division, Town of Barnstable, Paul Canniff, President, Wequaquet Lake Protective Association, and Gail Maguire, Vice President, Wequaquet Lake Protective Association. Coastal Systems

Program, School of Marine Science and Technology, University of Massachusetts Dartmouth. 24 pp.

Eichner, E.M. and T.C. Cambareri, 1992. Technical Bulletin 91-001: Nitrogen Loading. Cape Cod Commission, Water Resources Office, Barnstable, MA. Available at: http://www.capecodcommission.org/regulatory/NitrogenLoadTechbulletin.pdf

Eichner, E.M., T.C. Cambareri, G. Belfit, D. McCaffery, S. Michaud, and B. Smith, 2003. Cape Cod Pond and Lake Atlas. Cape Cod Commission. Barnstable, MA.

Eichner, E.M., T.C. Cambareri, G. Belfit, D. McCaffery, S. Michaud, and B. Smith. 2003. Cape

Eichner, E.M., T.C. Cambareri, K. Livingston, C. Lawrence, B. Smith, and G. Prahm, 1998. Cape Coastal Embayment Project: Interim Final Report. Cape Cod Commission, Barnstable, MA.

Eichner, E.M., T.C. Cambareri, K. Livingston, C. Lawrence, B. Smith, and G. Prahm. 1998. The Cape Cod Coastal Embayment Project. Cape Cod Commission, Water Resources Office, Barnstable, MA.

Ellis, M.Y., 1978. Coastal Mapping Handbook, Department of the Interior, U.S. Geological Survey and U.S. Department of Commerce, National Ocean Service and Office of Coastal Zone Management, U.S. GPO, Washington, D.C.

Fischer, H. B., List, J. E., Koh, R. C. Y., Imberger, J., and Brooks, N. H. (1979). *Mixing in inland and coastal waters*. Academic. San Diego.

FitzGerald, D.M., 1993. “Origin and Stability of Tidal Inlets in Massachusetts.” In: Coastal and Estuarine Studies: Formation and Evolution of Multiple Tidal Inlets, Volume 29, Symposium on Hydrodynamics and Sediment Dynamics of Tidal Inlets (D. G. Aubrey and G.S. Geise, eds.). American Geophysical Union, Washington, D.C. pp. 1-61.

Frimpter, M.H., J.J. Donohue, M.V. Rapacz. 1990. A mass-balance nitrate model for predicting the effects of land use on groundwater quality. U.S. Geological Survey Open File Report 88:493.

Geise, G.S., 1988. “Cyclical Behavior of the Tidal Inlet at Nauset Beach, Massachusetts: Application to Coastal Resource Management.” In: Lecture Notes on Coastal and Estuarine Studies, Volume 29, Symposium on Hydrodynamics and Sediment Dynamics of Tidal Inlets (D. Aubrey and L. Weishar, eds.), Springer-Verlag, NY, pp. 269-283.

Hägg, H.E., C. Humborg, D.P. Swaney and C.M. Mörth. 2011. Riverine nitrogen export in Swedish catchments dominated by atmospheric inputs. Biogeochemistry. DOI: 10.1007/s10533-011-9634-7

Hamersley, R.M. and B. Howes, 2004. Nitrogen Fluxes and Mitigation Strategies in the Audubon Skunknett River Wildlife Sanctuary. Report to the Town of Barnstable

Hampson, G.R. 1989. A REMOTS Survey of Buzzards Bay with Ground Truth Verification, EPA Report Region I Water Management Division, CR-8142976-01

Hampson, G.R., E.T. Moul. 1978. No. 2 Fuel Oil Spill in Bourne, Massachusetts: Immediate Assessment of the Effects on Marine Invertebrates and a 3-Year Study of Growth and Recovery of a Salt Marsh. Journal of Fisheries Research Bd. Canada 35(5):731-744

Harbaugh, A.W. and McDonald, M.G., 1996. User’s Documentation for MODFLOW-96, an update to the U.S. Geological Survey Modular Finite-Difference Ground-Water Flow Model: U.S. Geological Survey Open-File Report 96-485, 56p.

Harbaugh, A.W., Banta, E.R., Hill, M.C., and McDonald, M.G., 2000, MODFLOW-2000, The U.S. Geological Survey modular ground-water model—User guide to modularization concepts and the ground-water flow process: U.S. Geological Survey Open-File Report 00-92, 121 p.

Henderson, F. M., 1966. *Open Channel Flow*. Macmillan Publishing Company, New York. pp. 96-101.

Horsley Witten Group. 2012. South Sandwich Village: Draft Environmental Impact Report. Sandwich, MA.

Howes B., H.E. Ruthven, J. S. Ramsey, R. Samimy, D. Schlezinger, J. Wood, and E. Eichner. 2006a. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Centerville River System, Barnstable, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA. 172 pp.

Howes B., S. Kelley, J. S. Ramsey, E. Eichner, R. Samimy, D. Schlezinger, and P. Detjens. 2011. Massachusetts Estuaries Project Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Bass River System, Towns of Yarmouth and Dennis, Massachusetts. Massachusetts Department of Environmental Protection. Boston, MA. 222 pp.

Howes B., S. Kelley, J. S. Ramsey, E. Eichner, R. Samimy, D. Schlezinger, and P. Detjens. 2015. Massachusetts Estuaries Project Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Scorton Creek Estuarine System, Town of Sandwich, Massachusetts. Massachusetts Department of Environmental Protection. Boston, MA. 167 pp.

Howes B., S. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, E. Eichner (2009). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Parkers River Embayment System, Yarmouth, Massachusetts, Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B., S. W. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, E. Eichner (2003). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Stage Harbor, Sulphur Springs, Taylors Pond, Bassing Harbor, and Muddy Creek, Chatham, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B., S. W. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, E. Eichner (2006). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Three Bays, Barnstable, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B., S. W. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, E. Eichner (2006). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Three Bays, Barnstable, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B., S.W. Kelley, J.S. Ramsey, R. Samimy, D. Schlezinger, and E. Eichner. 2006b. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Three Bays, Barnstable, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA. 183 pp.

Howes B.L., E.M. Eichner, S. Kelley, J. S. Ramsey, R. Samimy, and D. Schlezinger. 2013. Massachusetts Estuaries Project Linked Watershed-Embayment Model to Determine

Critical Nitrogen Loading Thresholds for Fiddlers Cove and Rands Harbor Embayment Systems, Towns of Falmouth, Massachusetts. Massachusetts Department of Environmental Protection. Boston, MA. 174 pp.

Howes B.L., J. Ramsey, E.M. Eichner, R.I. Samimy, S. W. Kelley, D.R. Schlezinger (2005). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Oyster Pond System, Falmouth, MA. SMAST/DEP Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B.L., J. Ramsey, E.M. Eichner, R.I. Samimy, S. W. Kelley, D.R. Schlezinger (2005). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Three Bays System, Barnstable, MA. SMAST/DEP Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B.L., S.W. Kelley, J. S. Ramsey, R.I. Samimy, D.R. Schlezinger, E.M. Eichner (2007). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Rock Harbor Embayment System, Orleans, MA. SMAST/DEP Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B.L., S.W. Kelley, J. S. Ramsey, R.I. Samimy, E.M. Eichner, D.R. Schlezinger, (2007). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Namskaket Marsh Estuarine System, Orleans, MA. SMAST/DEP Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B.L., T. Ruthven, E. Eichner, R. Samimy, J. Ramsey, D. Schlezinger, P. Detjens (2015).

Howes, B., Kelley, S., Ramsey, J., Samimy, R., Eichner, E., Schlezinger, D., and Wood, J., 2004. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Popponesset Bay, Mashpee and Barnstable, Massachusetts. Commonwealth of Massachusetts, Department of Environmental Protection, Massachusetts Estuaries Project, 138 pp. + Executive Summary, 10 pp.

Howes, B.L. and C.T. Taylor. 1990. Nutrient Regime of New Bedford Outer Harbor: Infaunal Community Structure and the Significance of Sediment Nutrient Regeneration to Water Column Nutrient Loading. Final Technical Report to Camp, Dresser and McKee, Boston, Mass. for the City of New Bedford, Mass. and EPA. 140 pp.

Howes, B.L. and J.M. Teal. 1995. Nitrogen balance in a Massachusetts cranberry bog and its relation to coastal eutrophication. Environmental Science and Technology 29:960-974.

Howes, B.L. and J.M. Teal. 1995. Nitrogen balance in a Massachusetts cranberry bog and its relation to coastal eutrophication. *Environmental Science and Technology*. 29: 960-974.

Howes, B.L., D.D. Goehringer, N.P. Millham, D.R. Schlezinger, G.R. Hampson, C.D. Taylor and D.G. Aubrey. 1997. Nantucket Harbor Study: A quantitative assessment of the environmental health of Nantucket Harbor for the development of a nutrient management plan. Technical Report to the Town of Nantucket, pp. 110.

Howes, B.L., H.E. Ruthven, E.M. Eichner, J.S. Ramsey, R.I. Samimy, D.R. Schlezinger. 2008. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Lewis Bay System, Towns of Barnstable and Yarmouth, MA. SMAST/DEP Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA

Howes, B.L., J.S. Ramsey and S.W. Kelley, 2000. Nitrogen modeling to support watershed management: comparison of approaches and sensitivity analysis. Final Report to MA Department of Environmental Protection and USEPA, 94 pp. Published by MADEP.

Howes, B.L., J.S. Ramsey and S.W. Kelley, 2001. Nitrogen modeling to support watershed management: comparison of approaches and sensitivity analysis. Final Report to MA Department of Environmental Protection and USEPA, 94 pp. Published by MADEP.

Howes, B.L., R.I. Samimy and B. Dudley, 2003. Massachusetts Estuaries Project, Site-Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators Interim Report

Howes, B.L., R.I. Samimy, D.R. Schlezinger, S. Kelley, J. Ramsey, T. Ruthven, and E. Eichner, 2004. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Quashnet River, Hamblin Pond, and Jehu Pond, in the Waquoit Bay System of the Towns of Mashpee and Falmouth, MA. Massachusetts Estuaries Project Final Report, pp. 147.

Howes, D.L, D.D. Goehringer, 1996. Water Quality Monitoring of Falmouth's Coastal Ponds: Results from the 1994 and 1995 Seasons

IEP, Inc. and K-V Associates, Inc. 1989. Diagnostic/Feasibility Study of Wequaquet Lake, Bearse, and Long Pond. Prepared for Town of Barnstable, Conservation Commission. Sandwich and Falmouth, MA.

Jansson, M., Anderson, R., Berggren, H., Leonardson, L. Wetlands and Lakes as Nitrogen Traps, Ambio, vol. 23(6), pp. 320-325, 1994

Johnston, C. 1991. Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. *Critical Reviews in Environmental Control.* 21:491-565.

Jorgensen, B.B. 1977. The sulfur cycle of a coastal marine sediment (Limfjorden, Denmark). Limnology Oceanography, 22:814-832.

Kelley, S.W., Ramsey, J.R., Côté, J.M., Wood, J.D. (2001). "Tidal Flushing Analysis of Coastal Embayments in Chatham, MA" Applied Coastal Research and Engineering, Inc. report prepared for the Town of Chatham. 115 pp.

King, Ian P. (1996). "Users Guide to RMA2 Version 4.2." US Army Corps of Engineers – Waterways Experiment Station Hydraulics Laboratory.

King, Ian P., 1990. "Program Documentation - RMA2 - A Two Dimensional Finite Element Model for Flow in Estuaries and Streams." Resource Management Associates, Lafayette, CA.

Klump, J. and C. Martens. 1983. Benthic nitrogen regeneration. In: Nitrogen in the Marine Environment, (Carpenter & Capone, eds.). Academic Press.

Koppelman, L.E. (Ed.). 1978. The Long Island comprehensive waste treatment management plan. Vol II. Summary documentation report, Long Island Regional Planning Board, Hauppage, N.Y.

K-V Associates, Inc. and IEP, Inc. 1993. Shallow Pond Diagnostic - Feasibility Study. Prepared for Town of Barnstable, Conservation Commission. Falmouth and Barnstable, MA.

Lindeburg, Michael R., 1992. *Civil Engineering Reference Manual, Sixth Edition*. Professional Publications, Inc., Belmont, CA.

Massachusetts Department of Environmental Protection, 1999. DEP Nitrogen Loading Computer Model Guidance. Bureau of Resource Protection. Boston, MA. Available at: http://www.state.ma.us/dep/brp/dws/techtool.htm

Massachusetts Department of Revenue. June, 2009. Property Type Classification Codes, Non-arms Length Codes and Sales Report Spreadsheet Specifications. Division of Local Services, Bureau of Local Assessment. Boston, MA. 16 pp.

Massachusetts Department of Revenue. March, 2012. Property Type Classification Codes, Non-arm's Length Codes and Sales Report Spreadsheet Specifications. Prepared by the Bureau of Local Assessment. Boston, MA.

Massachusetts Department of Revenue. Property Type Classification Codes, Non-arm's Length Codes and Sales Report Spreadsheet Specifications. Prepared by the Bureau of Local Assessment. Boston, MA. 18 pp.

Massachusetts Estuaries Project Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Sandwich Harbor Estuary, Town of Sandwich, Massachusetts, Massachusetts Department of Environmental Protection. Boston, MA.

Massachusetts Water Resources Authority, 1983. Water supply study and environmental impact statement for the year 2020, Task I: Water demand projections. MWRA Report, Boston.

Masterson, J.P., Walter, D.A., and Savoie, Jennifer, 1997, Use of particle tracking to improve numerical model calibration and to analyze ground-water flow and contaminant migration, Massachusetts Military Reservation, western Cape Cod, Massachusetts: U.S. Geological Survey Water-Supply Paper 2482, 50 p.

Masterson, J.P., Walter, D.A., Savoie, J., 1996, Use of particle tracking to improve numerical model calibration and to analyze ground-water flow and contaminant migration, Massachusetts Military Reservation, western Cape Cod, Massachusetts: U.S. Geological Survey Open-File Report 96-214, 50 p.

Millham, N.P. and B.L. Howes, 1994a. Freshwater flow into a coastal embayment: groundwater and surface water inputs. Limnology and Oceanography 39: 1928-1944.

Millham, N.P. and B.L. Howes, 1994a. Freshwater flow into a coastal embayment: groundwater and surface water inputs. *Limnology and Oceanography* 39: 1928-1944.

Millham, N.P. and B.L. Howes, 1994b. Patterns of groundwater discharge to a shallow coastal embayment. Marine Ecology Progress Series 112:155-167.

Millham, N.P. and B.L. Howes, 1994b. Patterns of groundwater discharge to a shallow coastal embayment. *Marine Ecology Progress Series* 112:155-167.

Millham, N.P. and B.L. Howes. 1994. A comparison of methods to determine K in a shallow coastal aquifer. Groundwater. 33:49-57.

Millham, N.P. and B.L. Howes. 1994. Nutrient balance of a shallow coastal embayment: I. Patterns of groundwater discharge. Marine Ecology Progress Series 112:115-167.

Murphy, J. and J.P. Reilly, 1962. A Modified Single Solution Method for the Determination of Phosphate in Natural Waters. Analytica Chemica Acta, v. 27, p. 31-36

Nelson, M.E., S.W. Horsley, T.C. Cambareri, M.D. Giggey and J.R. Pinnette. 1998. Predicting nitrogen concentrations in groundwater- An analytical model. Focus Conference on Eastern Groundwater Issues, National Water Well Association, Stamford, CT.

Norton, W.R., I.P. King and G.T. Orlob, 1973. "A Finite Element Model for Lower Granite Reservoir", prepared for the Walla Walla District, U.S. Army Corps of Engineers, Walla Walla, WA.

O'Hara, C.J., Oldale, R.N., 1987. Geology, Shallow Structure, and Bedform Morphology, Nantucket Sound, Massachusetts. 1:125,000. United States Geological Survey Miscellaneous Field Studies Map MF 1911.

Oldale, R. N., 1992, Cape Cod and the Islands: The geologic history: East Orleans, MA, Parnassus Imprints, 208 p.

Oldale, R.N. 1974a. Geologic Map of the Hyannis Quadrangle, Barnstable County, Cape Cod, Massachusetts. US Geological Survey Map GQ-1158. US Geological Survey, Reston, VA.

Oldale, R.N. 1974b. Geologic Quadrangle Maps of the United States, Geologic Map of the Dennis Quadrangle, Barnstable County, Cape Cod, Massachusetts. US Geological Survey Map GQ-1114. US Geological Survey, Washington, DC.

Pleasant Bay Technical Advisory Committee, and Ridley & Associates, Inc., 1998. Pleasant Bay resource management plan. Report to the Pleasant Bay Steering Committee, 158 pp + app.

Pollock, D.W., 1994, User's guide for MODPATH/ MODPATH-PLOT, version 3—A particle tracking post-processing package for MODFLOW, the U.S. Geological Survey finite-difference ground-water flow model: U.S. Geological Survey Open-File Report 94-464, 234 p.

Pollock, D.W., 1994. User's Guide to MODPATH/MODPATH_PLOT, version 3 – A particle tracking post-processing package for MODFLOW, the U.S. Geological Survey modular three dimensional finite-difference ground-water-flow-model: U.S. Geological Survey Open-File Report 94-464, [variously paged].

Poucher, S. and L. Coiro. 1997. Test Reports: Effects of low dissolved oxygen on saltwater animals. Memorandum to D.C. Miller. U.S. Environmental Portection Agency, Atlantic Ecology Division, Narragansett, Rhode Island. (Cited in New York State Aquatic Fact Sheet on Ambient Water Quality Value for Protection of Aquatic Life, 2008).

Ramsey, J.S., B.L. Howes, N.P. Millham, and D. Bourne. 1995. Hydrodynamic and water quality study of West Falmouth Harbor, Falmouth MA. Aubrey Consulting Inc. Technical Report for Town of Falmouth, pp. 81.

Ramsey, J.S., B.L. Howes, S.W. Kelley, and F. Li (2000). "Water Quality Analysis and Implications of Future Nitrogen Loading Management for Great, Green, and Bournes Ponds, Falmouth, Massachusetts." Environment Cape Cod, Volume 3, Number 1. Barnstable County, Barnstable, MA. pp. 1-20.

Ramsey, John S., Jon D. Wood, and Sean W. Kelley, 1999. "Two Dimensional Hydrodynamic Modeling of Great, Green, and Bournes Ponds, Falmouth, MA." Applied Coastal Research and Engineering, Inc. report prepared for the Town of Falmouth and Horsley & Witten, Inc. 41 pp.

Rhoads, D.C. and J.D. Germano. 1986. Interpreting long-term changes in benthic community structure: a new protocol. Hydrobiologia 142:291-308

Robertson, W.D., Cherry, J.A., Sudicky, E.A., 1991. Groundwater Contamination from Two Small Septic Systems on Sand Aquifers. Groundwater 29, 82-92.

Robertson, W.D., S.L. Schiff, and C.J. Ptacek. 1998. Review of Phosphate Mobility and Persistence in 10 Septic System Plumes. Ground Water, 36(6):1000-1010.

Ryther, J.H., and W.M. Dunstan. 1971. Nitrogen, phosphorous and eutrophication in the coastal marine environment. Science, 171:1008-1012.

Sanders, H.L. 1960. Benthic studies in Buzzards Bay III. The structure of the soft-bottom community. Limnology and Oceanography 5:138-153.

Sanders, H.L., J.F. Grassle, G.R. Hampson, L.S. Morse, S. Gameprice, and C.C Jones. 1980. Anatomy of an oil spill: Long-term effects from the grounding of the barge *Florida* off West Falmouth, MA. Journal of Marine Research 38:265-380.

Saunders, D.L. and J. Kalff. 2001. Nitrogen retention in wetlands, lakes and rivers. *Hydrobiologica.* 443: 205-212.

Scheiner, D. 1976. Determination of ammonia and Kjeldahl nitrogen by indophenol method. Water Resources 10: 31-36.

Shalowitz, A.L., 1964. Shore and Sea Boundaries--with special reference to the interpretation and use of Coast and Geodetic Survey Data. U.S. Department of Commerce Publication 10-1, Two Volumes, U.S. GPO, Washington, D.C.

Smith, K. 1999. Salt Marsh Uptake of Watershed Nitrate, Mashapaquit Creek Marsh, West Falmouth Harbor, Falmouth, Cape Cod, Massachusetts. Masters Thesis, Boston University Department of Earth Sciences, Boston, pp. 1-76.

Smith, K.N. and B.L. Howes. Manuscript. Attenuation of watershed nitrogen by a New England salt marsh: a buffer for cultural eutrophication of coastal waters.

Smith, R.L., B.L. Howes and J.H. Duff. 1991. Denitrification in nitrate-contaminated groundwater: occurrence in steep vertical geochemical gradients. Geochimica Cosmochimica Acta 55:1815-1825.

Stearns and Wheler. 2001. Wastewater Facilities Plan and Final Environmental Impact Report for the Town of Falmouth, Massachusetts. Hyannis, MA.

Taylor, C.D. and B.L. Howes, 1994. Effect of sampling frequency on measurements of seasonal primary production and oxygen status in near-shore coastal ecosystems. Marine Ecology Progress Series 108: 193-203.

Thieler, E.R., J.F. O'Connell, C.A. Schupp, 2001. The Massachusetts Shoreline Change Project: 1800s to 1994. Technical Report, 60 p.

Tian, Y.Q., J.J. Wang, J. A. Duff, B.L. Howes and A. Evgenidou. 2009. Spatial patterns of macrobenthic communities in shallow tidal embayments and association with environmental factors. Journal of Environmental Management 44:119-135.

Toet, S., Van Logtestijn, R.S.P., Kampf, R., Schreijer, M., and J.T.A. Verhoeven. 2005. The effect of hydraulic retention time on the removal of pollutants from sewage treatment plan effluent in a surface-flow wetland system. *Wetlands* 25(2), pp. 375–391.

Toet, S., Van Logtestijn, R.S.P., Kampf, R., Schreijer, M., and J.T.A. Verhoeven. 2005. The effect of hydraulic retention time on the removal of pollutants from sewage treatment plan effluent in a surface-flow wetland system. *Wetlands*. 25(2): 375–391.

U.S. Army Corps of Engineers (1964). "Beach Erosion Control Report on Cooperative Study of Falmouth, Massachusetts." Headquarters, Department of the Army, Office of the Chief of Engineers, Washington, D.C.

U.S. Army Corps of Engineers, New England Division, Tidal Flood Profiles, New England Coastline, September 1988.

US Army, Engineer Research and Development Center, Waterways Experiment Station, Coastal and Hydraulics Laboratory, Users Guide To RMA4 WES Version 4.5, June 05, 2001.

USGS web site for groundwater data for Massachusetts and Rhode Island: http://ma.water.usgs.gov/ground_water/ground-water_data.htm

Van de Kreeke, J., 1988. “Chapter 3: Dispersion in Shallow Estuaries.” In: Hydrodynamics of Estuaries, Volume I, Estuarine Physics, (B.J. Kjerfve, ed.). CRC Press, Inc. pp. 27-39.

Walter, D.A. and A.T. Whealan, 2005. Simulated Water Sources and Effects of Pumping on Surface and Ground Water, Sagamore and Monomoy Flow Lenses, Cape Cod, Massachusetts. US Geological Survey Scientific Investigations Report 2004-5181, 85 p.

Walter, D.A. and Whealan, A.T. 2005. Simulated Water Sources and Effects of Pumping on Surface and Ground Water, Sagamore and Monomoy Flow Lenses, Cape Cod, Massachusetts. US Geological Survey Scientific Investigations Report 2004-5181, 85 p.

Watershed. *Water Resources Research*. 27(11): 2929-2939.

Weiskel, P.K. and B.L. Howes, 1991. Quantifying Dissolved Nitrogen Flux Through a Coastal Watershed. Water Resources Research, Volume 27, Number 11, Pages 2929-2939.

Weiskel, P.K. and B.L. Howes, 1992, Differential Transport of Sewage Derived Nitrogen and Phosphorous through a Coastal Watershed. Environmental Science and Technology, Volume 26, No. 2, pp. 352 - 360

Weiskel, P.K. and B.L. Howes. 1991. Quantifying Dissolved Nitrogen Flux Through a Coastal

Weiskel, P.K. and B.L. Howes. 1992. Differential Transport of Sewage-Derived Nitrogen and Phosphorous through a Coastal Watershed. *Environmental Science and Technology*. 26: 352-360.

Wilhelm, S.R., S.L. Schiff, and W.D. Robertson. 1996. Biogeochemical Evolution of Domestic Waste Water in Septic Systems: 2. Application of Conceptual Model in Sandy Aquifers. Ground Water, 34(5):853-864.

Wood, J.D., J.S. Ramsey, and S. W. Kelley, 1999. “Two-Dimensional Hydrodynamic Modeling of Barnstable Harbor and Great Marsh, Barnstable, MA.” Applied Coastal Research and Engineering, Inc. report prepared for the Town of Barnstable. 28 pp.

Wright-Pierce. 2012. Sandwich, Massachusetts Comprehensive Water Resources Management Plan, Needs Assessment. Andover, MA.

Zimmerman, J.T.F., 1988. “Chapter 6: Estuarine Residence Times.” In: Hydrodynamics of Estuaries, Volume I, Estuarine Physics, (B.J. Kjerfve, ed.). CRC Press, Inc. pp. 75-84.